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Effects of the Cessation of Sewage Sludge Dumping at the 12-Mile Site

12-Mile Dumpsite Symposium
Long Branch, New Jersey, June 1991

Anne L. Studholme
John E. O'Reilly
Merton C. Ingham (editors)

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NOAA Technical Report NMFS 124

A Technical Report of the *Fishery Bulletin*

Effects of the Cessation of Sewage Sludge Dumping at the 12-Mile Site

Proceedings of the 12-Mile Dumpsite Symposium
Ocean Place Hilton Hotel, Long Branch, New Jersey
18–19 June 1991

Anne L. Studholme
John E. O'Reilly
Merton C. Ingham (editors)

Sponsored by:

Environmental Processes Division
Northeast Fisheries Science Center
National Marine Fisheries Service
U.S. Dept. of Commerce, NOAA
Woods Hole, Massachusetts

October 1995

U.S. Department of Commerce
Seattle, Washington

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PREFACE

This study owes its inception to the wisdom and experience of the staff of the Northeast Fisheries Science Center who, after several decades of surveys in the New York Bight, recognized a unique opportunity to capitalize on the decision to stop ocean dumping of sewage sludge and designed an innovative field study to evaluate effects on living marine resources and their habitats. For decades ocean dumping was viewed as a cheap and effective means for disposal of wastes generated by urbanized coastal areas. Even after the 12-mile site was closed, sewage sludge continued to be dumped at Deepwater Dumpsite 106. The 6-mile site off the New Jersey coast is still used as a dumpsite for dredged material from New York Harbor areas. Discussions continue on the propriety of using the deep ocean spaces for disposal of a variety of material including low level radioactive wastes. Consequently, managers are still faced with critical decisions in this area. It is to be hoped that the results from the 12-mile study will provide the necessary information on which these managers can evaluate future risks associated with ocean waste disposal.

INTRODUCTION

In anticipation of the closure of the 12-mile sewage sludge dumpsite in the New York Bight, in July 1986 the Northeast Fisheries Science Center of the National Marine Fisheries Service (NMFS) proposed a major interdisciplinary program to evaluate the effects of the removal of a major source of contaminants on the region's marine resources and their habitats (Studholme, 1988). As described in the plan for study prepared by the Environmental Processes Division (EPD, 1988), for over six decades, as many as 200 municipalities in New York and New Jersey had used the Bight as a relatively inexpensive disposal site, not only for sewage sludge but also for dredge materials and other wastes (Fig. 1). Paradoxically, although the number of municipalities using the 12-mile site had decreased to as few as nine in recent years, the amount of sludge dumped had increased from approximately 4.2×10^6 wet tons in 1974 to 8.3×10^6 wet tons in 1983 primarily because of the upgrading of waste treatment facilities in the New York and New Jersey areas (Fig. 2). Thus, during the 1980's, the 12-mile site received larger volumes of sludge than any other site in this country or abroad (Norton and Champ, 1989).

The decision by the U.S. Environmental Protection Agency (EPA) in 1985 to discontinue use of the site was the culmination of over 15 years of controversy. Beginning in 1970, the U.S. Food and Drug Administration (FDA) acting under the auspices of the National Shellfish Sanitation program closed a 380 km^2 area around the dumpsite to commercial shellfishing following the discovery of elevated levels of coliform bacteria in sediment and shellfish (Fig. 1; Verber, 1976). Furthermore, there was evidence that sewage sludge dumped at the site was a significant source of contaminants to the adjacent Christiaensen Basin and Hudson Shelf Valley (HSV). As a result, EPA, acting under an amendment to the Marine Protection, Research, and Sanctuaries Act (MPRSA), denied any further permits for dumping after December 1981 (Erdheim, 1985; Santoro, 1987).

Although both New York and New Jersey brought suit to continue use of the 12-mile site, EPA, in April 1985, denied further requests for its redesignation, attributing a number of ecological effects to the dumping of municipal sludge including 1) the introduction of a variety of pathogens, 2) elevated levels of heavy metals and organic compounds, 3) reduced catch of fish, 4) reduced levels of bottom dissolved oxygen, 5) significant changes in benthic communities, particularly the appearance of polychaete species indicative of pollutant stress, 6) a variety of sublethal effects including decreased reproductive potential, increased incidence of fish disease, and decreased survival of offspring, and 7) increased carbon and nutrients contributing to plankton blooms and potential hypoxia (Anderson[1]). Subse-

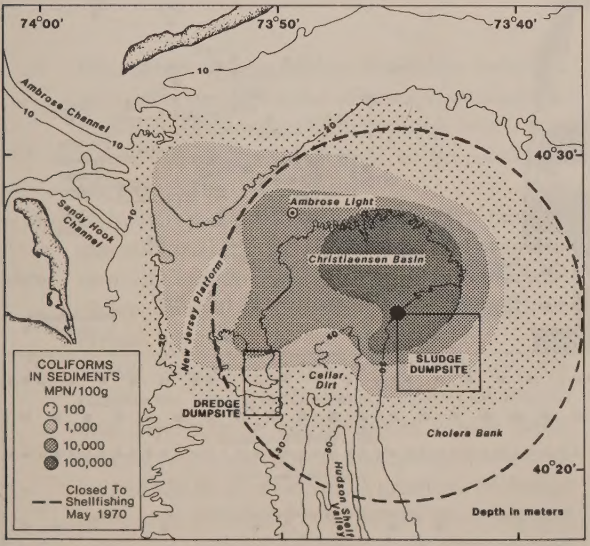

Figure 1

Location of the 12-mile dumpsite, dredge materials dumpsite, and area closed to commercial shellfishing in the New York Bight (after Verber, 1976). MPN=most probable number.

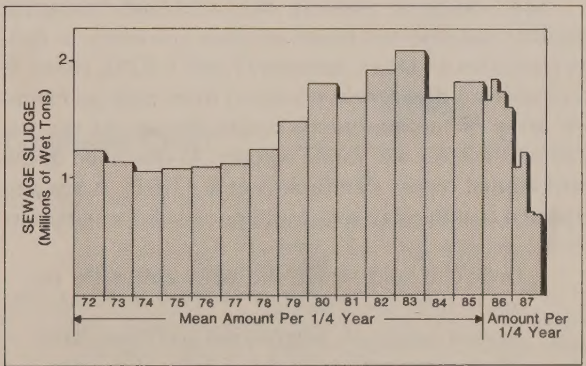

Figure 2

Estimated volumes of sewage sludge dumped in the New York Bight. Annual means for 1972–85; quarterly totals for 1986–87 (after Swanson et al., 1985; Santoro, 1987). Sources of data: Suszkowski and Santoro, 1986, and USEPA Region II, Marine and Wetlands Protection Branch, New York, NY, 1988.

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[1] Statement of Dr. Peter Anderson at the Public Hearing on the Tentative Denial to Redesignate the 12-Mile Sewage Sludge Dumpsite, Monmouth College, NJ, 18 June 1984.

quently, in March 1986, a scheduled phasing out of sludge disposal was begun and the site was closed by December 1987 (Santoro, 1987).

Apart from surveys made by NMFS, EPA, and FDA before and after closure of the Philadelphia dumpsite (Devine and Simpson, 1985) little information existed about "recovery" of disposal sites. The Philadelphia site was in a highly dispersive system that precluded any substantial sludge accumulation and after only 4 years, microbial contamination had returned to background levels (Lear and O'Malley, 1983). In sharp contrast, areas adjacent to the 12-mile site are highly depositional and there was uncertainty whether cessation of dumping would eventually result in significant changes in water and sediment chemistry, microbial contamination, benthic community structure, distribution and abundance of demersal finfish, changes in food habits, or other ecological effects (Swanson et al., 1985).

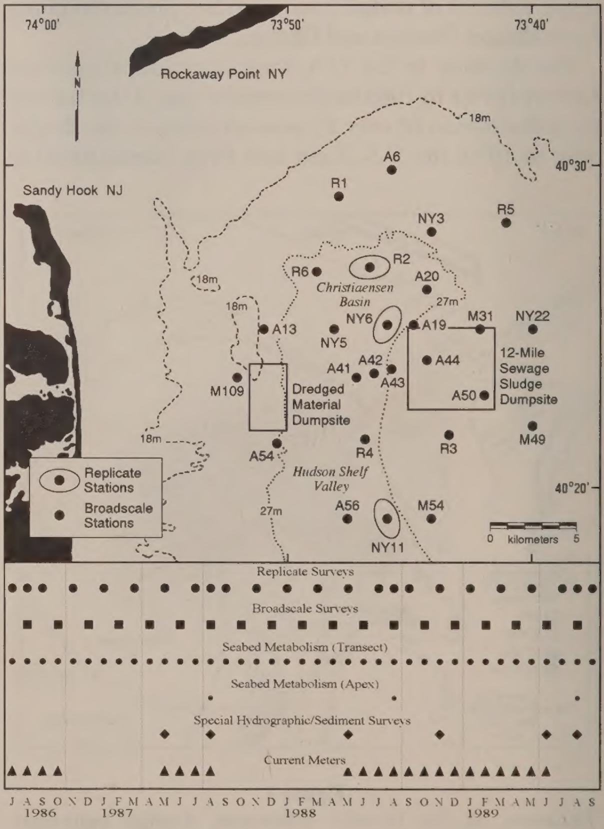
A number of considerations had to be taken into account in the design of the study (Studholme et al., 1991). The first was the long history of degradation in the Bight that precluded the availability of a predumping baseline. The second difficulty stemmed from separating sludge effects from those attributable to other major sources of pollution entering the Bight, including contributions from the Hudson-Raritan plume and the dredged material disposed at a site midway between the sludge dumpsite and the New Jersey coast, i.e. the 6-mile site. The survey design addressed these problems by including two types of complementary surveys that would provide both repeated measurements at three stations representing a gradient of sludge influence (replicate surveys) and a more extensive descriptive survey (broadscale survey) covering nearly 350 km^2 of the Bight (Fig. 3).

A wide range of physical, chemical, and biological variables was selected based on their relevance to fishery resources and their habitats (Table 1; EPD, 1988). It was expected that results obtained from such an extensive array of measurements would document the degree to which the site would recover. As described in the third annual report (Studholme et al., 1991), it was also expected that these results would provide information to

1. Define if and when the area could be reopened for shellfishing;
2. Assess changes in distribution and abundance of resource species in the vicinity of the site;
3. Better resolve sediment resuspension and transport of sludge components out of the Christiaensen Basin;
4. Quantify areal response to changes in pollutant loading in order to permit prediction of response rates for future waste disposal sites; and
5. Provide a basis for comparison to evaluate EPA's decision to activate the 106 Deepwater Dumpsite.

A number of changes at the site were anticipated during and following closure (EPD, 1988; Studholme et al., 1991). The following lists some of the anticipated changes formulated as working hypotheses:

1. Dispersion of sewage sludge and cleansing of the Christiaensen Basin would be influenced by local windfield conditions and resultant changes in bottom-water circulation in the upper HSV.
2. Changes in water and sediment chemistry would occur, including reduction of sediment trace metals in sludge depositional areas. However, concentrations of certain organic compounds (e.g. PCB's) might be more stable. Seabed oxygen consumption and nutrient regeneration rates should be reduced as benthic community metabolism and organic loading decrease.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Rockaway Point NY
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Christiaensen Basin
A20
NY6
A19
M31
NY22
A13
NY5
A44
12-Mile Sewage Sludge Dumpsite
A41
A42
A43
M109
Dredged Material Dumpsite
A50
R3
M49
R4
A54
Hudson Shelf Valley
Replicate Stations
Broadscale Stations
A56
M54
NY11
0 kilometers 5
Replicate Surveys
Broadscale Surveys
Seabed Metabolism (Transect)
Seabed Metabolism (Apex)
Special Hydrographic/Sediment Surveys
Current Meters
J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J J A S
1986
1987
1988
1989

Figure 3

Replicate, broadscale, and seabed metabolism stations and survey schedule for the 12-mile dumpsite study.

Table 1

Variables measured during the 12-mile dumpsite study.

Habitat		
Water	Sediment	Biota
Bottom water	Chemistry	Resource species
Dissolved oxygen (R,B[1])	Heavy metals (R,B)	Distribution/abundance (R,B)
Temperature (R,B)	Organic contaminants (R,B)	Diet (R)
Salinity (R,B)	Sulfide, pH profiles (R)	Winter flonder
pH (R,B)	Redox potential (R,B)	Red hake
Sulfide (R,B)	Sediment BOD[2] (R)	American lobster
Nutrients (R,B)	Chlorophyll pigments (R,B)	Gross pathology (R)
Turbidity	Total organic carbon (R,B)	Winter flounder
Water column	Characteristics	American lobster
Temperature	Grain size (R,B)	Tissue organics (R)
Salinity (CTD)	Erodibility	Winter flounder
Oxygen	Rates	American lobster
Current measurements	Seabed oxygen consumption	Migration (tagging) (B)
(moored meters)	Sedimentation	Winter flounder
		American lobster
		Benthos
		Macrofauna abundance/diversity (R,B)
		Meiofauna abundance/diversity (R,B)
		Bacteria - sediments
		Fecal & total coliform (R,B)
		C. perfringens (R)
		Vibrio spp. (R)
		Total count (R)

[1] R = replicate survey; B = broadscale survey.
[2] BOD = biological oxygen demand.

3. Microbial concentrations should be reduced, and bacteria indicative of sewage contamination should decrease to acceptable levels that would permit shellfish beds to be reopened for harvesting.
4. In heavily polluted areas, numbers of benthic macrofaunal species should increase; populations of *Capitella capitata*, an opportunistic polychaete found in disturbed areas, should decrease.
5. While dumping continued, abundance, distribution, and species composition of finfish and invertebrate communities would differ among areas which were bathymetrically similar but which represented a gradient of sludge influence. Following cessation of dumping and expected shifts in sediment contaminants and benthic forage species, these spatial differences should be reduced.

These as well as other working hypotheses were addressed during the three-year field study. The results were presented at a symposium in Long Branch, New Jersey, in June 1991 and form the basis for the papers presented in this volume.

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Introduction

ROBERT A. MURCHELANO

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On behalf of the Environmental Processes Division, Merton Ingham, Anne Studholme, John O'Reilly, and most importantly all the members of the Environmental Processes Division that conducted this study for the past three years, I welcome you to the Symposium. This is a milestone which was planned when we began this investigation in 1985. We thought it would be the legitimate outcome of three years of intensive activity at the 12-mile dumpsite. It is fitting that we discuss our findings, with you, help you interpret what we have done, and give you an opportunity for questions.

A little history of the genesis of this particular study is in order. In 1986, a NOAA document on the most important ocean pollution issues identified ocean dumping as one of the top four priorities. Dr. Jack Pearce would be among the first to remember that ocean dumping research in the Middle Atlantic Bight began many years ago. In 1968, the U.S. Army Corps of Engineers funded research that was conducted at the former Fish and Wildlife Service Laboratory at Sandy Hook, New Jersey; that research continued until about 1971. Subsequent to that, the Marine EcoSystem Analysis (MESA) Project of NOAA funded ocean dumping research from 1973 to 1980.

The location of Sandy Hook provides convenient access to the area called the New York Bight, and if you look at the history of the laboratory in terms of involvement in habitat or environmental science, it certainly focused on the Bight apex and coastal problems. Ocean dumping—an activity that took place for over 60 years—its magnitude, and the fact that it was to stop, provided a unique opportunity to look at the course of events biologically, chemically, and physically. That is really what made the study an imperative for us.

I have given you a little perspective as to why; let me now tell you who. NOAA/NMFS is the principal, but not the only organization involved in the 12-Mile Dumpsite Study. I would like to express our sincere appreciation to those who were involved, including:

1. The New Jersey Department of Environmental Protection (DEP)—specifically Dr. Robert Tucker. All he did for us for three years must be acknowledged and is appreciated. In particular, DEP's help enabled us to construct a Gas Chromatograph/Mass Spectrometer Laboratory and conduct organic analyses.
2. The U.S. Environmental Protection Agency (EPA) at Narragansett, Rhode Island—particularly Dr. Wayne Davis. Dr. Davis evaluated the possibility of sludge resuspension, a fundamental factor affecting transport of the sludge reservoir.
3. Dr. Irwin Katz at the U.S. EPA Laboratory in Edison, New Jersey. Dr. Katz quantified *Clostridium perfringens*, a bacterial indicator of the presence of sewage. He conducted an important bacteriological study that has helped enormously in defining the distribution of sewage sludge in the Bight.
4. Mr. Jack Gaines of the U. S. Food and Drug Administration (FDA) at Davisville, Rhode Island. He addressed the fundamental question of whether bivalve molluscs at the 12-mile dumpsite that are used for food can be safely put on the market again. A very basic question was to determine whether these shellfish have been purified so that they can be marketed.
5. Dr. Richard Bopp of Lamont-Doherty Earth Observatory. He looked at the distribution of organic chemical contaminants in sediments to determine whether they are waterborne or airborne and where they are transported.

Without these collaborators our efforts would have been missing significant interpretive elements. They have added essential components to the overall study.

All three branches of the Environmental Processes Division were involved in the study. Oceanographic data were acquired by the Physical Oceanography Branch led by Dr. Merton Ingham. The Branch conducted an oceanographic survey of the area, a prerequisite for the chemistry and biology. The information the Branch provided made it possible to determine whether the hydrography of the area was characteristic, or whether it deviated substantially from normal, a fundamental consideration before interpretation of other events could be made.

The Physical Oceanography Branch also determined the direction of water movement, particularly near the bottom. With the use of current meters they determined where materials could be transported by water and what related events, i.e. atmospheric and meteorologic, could influence water movement.

The Chemical Processes Branch, under the direction of Mr. John O'Reilly, evaluated metal concentrations in sediments and determined whether concentrations and distributions of metals are persistent. One of the issues cited as a reason for closure of the site related to metals and their elevated concentration and pervasive distribution.

The acquisition of the Gas Chromatograph/Mass Spectrometer made it possible to conduct analyses for persistent toxic chemicals and determine their prevalence and importance as contaminants causing biological effects. After standardizing procedures, we have just begun to evaluate specimens that came from the New York Bight—two species in particular: winter flounder and lobster.

We have studied sediment biogeochemistry and measured changes in redox potential to determine how healthy the sediments are and whether they are aerobic or anaerobic. Redox potential is a useful integrator of a number of chemical and biological activities.

The Environmental Assessment Branch, led by Anne Studholme, looked at the changes in the abundance of various species, both finfish and shellfish—the megabenthos. They looked at patterns of movement and migration of winter flounder, an important recreational species which is also useful as a biological indicator. They looked at disease prevalence in winter flounder, changes in diets of fish and shellfish, and changes in the composition of the macrobenthos.

From the outset we proposed a study design that would make it possible to use this large data set in a matrix fashion to interrelate parameters in the scale that was necessary for such a large study.

This investigation is not complete. Three years of study is a short period of time when compared with the 64 years of ocean dumping. There may be valid reasons to return to the area at some future time and make additional biological, chemical or physical measurements. What that intervening time is remains to be seen. First we must review our available data.

A point to note is that we have a massive data set that at present has been examined only superficially. We have only looked at changes during the first year and a half after dumping stopped. The process of synthesizing the data will continue.

Finally, I have to acknowledge that the Environmental Processes Division stayed the course for this study for over three years despite numerous difficulties within the Fisheries Service. The last three years, as you all know, have been traumatic ones with respect to federal budgets, federal climates, and personnel changes. We have had seven changes within NOAA of either the Director of NOAA or the Director of the Fisheries Service. We have had the impact of Gramm-Rudman legislation. We have sustained incremental budgetary cuts each year. We started with a staff of 82 at Sandy Hook and ended with 47. Yet 39 monthly cruises were completed successfully. We acquired our samples and stayed the course!

With that by way of background, I would like to introduce our first guest speaker. I am honored to have Mr. John Keith of the Department of Environmental Protection of the State of New Jersey as our keynote speaker. John is Assistant Commissioner for Environmental Management and Control for the State of New Jersey Department of Environmental Protection. He directs the activities of some 1,500 personnel in the Divisions of Environmental Quality, Water Resources, and Solid Waste Management. He has a bachelor's degree in civil engineering from the University of Michigan and a master's degree in environmental engineering from New Jersey Institute of Technology. He has over 19 years experience in environmental science, acquired during his tenure positions in state government and industry.

It is my pleasure this morning to present as our first speaker, Mr. John Keith.

Keynote Address

JOHN KEITH*

Assistant Commissioner
New Jersey Department of Environmental Protection
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It is particularly appropriate that we have this conference on the effects of cessation of ocean dumping. In New Jersey we are in the midst of major changes in policy directions, and sometimes we need to stop and think about what is the effect of what we are doing—what is the effect of what we have done.

I would like to focus my remarks today on perceptions versus reality. As I am sure you are all aware, we in government deal to a large extent with perceptions. What does the public perceive as the public good or the problems that government needs to address? What gets votes? What causes political controversy or creates a lot of press? Those are all based on perceptions.

You in this room are scientists. You deal with reality. We in the Department of Environmental Protection find ourselves in the unusual circumstance of trying to balance perceptions versus reality: to address what the public, public leaders, and the press are concerned about from a perception point of view, and yet keep in mind what the reality is and what the real problems are.

I think that ocean dumping of sludge at the 12-mile site or the 106-mile site is a perfect example of perceptions versus reality. Ocean dumping of sludge started in 1924 at the 12-mile site, six miles off the coast. The perception then was very clear. The ocean is an unlimited disposal resource. There is no way that humans can affect the integrity of the ocean.

The reality, which we are going to study in this conference and we are going to continue to study as a result of ongoing research, is that we certainly can affect the ocean. We can kill or do enormous damage to portions of it.

The 12-mile site, located 12 miles southeast of Sandy Hook, comprises 6.6 square miles with water depths between 22 and 27 meters. Over the years ocean dumping of sludge and other material increased. By 1970 approximately 200 sewage treatment plants were dumping approximately 5 million wet tons of sewage sludge per year at the 12-mile site. Before passage of the Marine Protection Research and Sanctuaries Act (MPRSA) on 23 October 1972, there were almost no legal controls, either inside or outside the United States, of the dumping of wastes at sea. Under MPRSA, the U. S. Environmental Protection Agency (EPA) was authorized to administer and enforce the Ocean Dumping Code, an ocean dumping program, and to issue permits. Still ocean dumping went on, and there was not much work to determine what the effects were.

In 1977, President Carter signed into law an amendment to the Act, legislating that by 31 December 1981, there would be an end to ocean dumping of sewage sludge that could cause unreasonable environmental impact. The problem, of course, was a definition of what constitutes an unreasonable ennvironmental impact. The public had not perceived that there was a real problem. Therefore, the dumping went on.

There was a famous court case where New York City sued EPA in Federal court arguing that Congress had only barred dumping which unreasonably degrades a marine environment. The City further argued that there was no evidence that the dumping at the 12-mile site had indeed unreasonably degraded the environment. So the ocean dumping at the 12-mile site went on.

In 1984, EPA designated the 106-mile site, and in April of 1985, the Agency took formal action to finally close the 12-mile site. In 1985, EPA imposed the following schedule on New Jersey ocean dumpers: that by 1986, 25 percent of the sludge dumped by the six New Jersey ocean dumpers would be moved out to the 106-mile site, and that by December 1987 all of the New Jersey sludge would be moved out to the 106-mile site.

Of course, the 106-mile site had not been evaluated to determine what the impact would be on the ecological system out there. True, it is a much bigger site (a hundred square miles) and in much deeper water (1,500 to 2,700 meters). But still, had anybody evaluated the impact? Basically the perception was that it had to be better than the 12-mile site.

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We have made much progress since 1986 when the 106-mile site began operation. In advance of State or Federal legislation, New Jersey's six ocean dumping authorities were required in the spring of 1989 to develop long-term and interim land-based sludge management plans that would handle all sewage sludge after 17 March of this year. In July 1988, Governor Kean signed the Ocean Dumping Elimination Act, prohibiting all New Jersey municipal treatment works from disposing sludge in the ocean after 17 March 1991.

Over the last several years the Department of Environmental Protection, interested parties, environmental groups, and most notably the sewer plants and the sewer authorities have worked very hard in cooperation with the courts and EPA to end ocean dumping of all sludge from New Jersey. It certainly was a great day on 17 March when we could announce that indeed all ocean dumping of New Jersey sludge had ended.

There was some perception that the problem has been solved. Unfortunately, as I am sure most of us are all too painfully aware, the sludge problem has not been solved. And I would like to get back to where we are going with sludge and address some of both perception and reality problems that we now have with sludge management.

Let me turn back to the idea of perceptions versus reality. Initially ocean dumping was thought to be good; gradually the perception shifted that ocean dumping of sludge was bad. But, what indeed were the actual effects of this practice; was it indeed good or bad? That is what I am hoping to hear at this conference and from the research discussed here. It is only by conducting this type of study that we are going to learn in reality what makes good policy as opposed to only addressing the perceptions of the public or the politicians.

Let me review some of the perception problems we have had with dumping the sludge at the 12-mile site. In 1976 there were hypoxic conditions offshore resulting in a widespread fish kill. The perception was originally that nutrient enrichment of near-coastal waters was attributable to sludge dumping at the 12-mile site. We may never be able to determine the causes or to what extent the hypoxic event was attributable to sludge dumping. Certainly sludge dumping deserves a portion of the blame. But for the 1976 hypoxia event the probable reason, as far as the Department can determine, was a continental shelf-wide bloom of *Ceratium tripos*, a species native to the northern oceans. Quiescent summer conditions with strong and persistent thermoclines in the ocean resulted in benthic hypoxia and subsequent fish kills.

In 1984, 1985, and again in 1986, indicator coliform bacteria in the near-shore coastal areas of Monmouth and Ocean Counties exceeded recreational bathing guidelines for our public health criteria. The public attributed this to onshore intrusions of sludge discharged at the 12-mile site. We believe now that the probable source of most, though not all, of that problem was storm-water discharge in the immediate vicinity of the bathing areas.

Storm water, as you are aware, picks up waste materials from roads and other areas, and, with combined sewer problems and improper connections of the storm sewers, the runoff during the storms can result in closure of beaches.

In 1986, there were widespread reports of materials washing up on our beaches: hypodermic syringes, tampon applicators, and other materials that we certainly find unsightly and unpleasant. It caught a lot of press. Once again, sludge dumping at the 12-mile site was the immediate target of concern. We believe now that, like the bacterial contamination, combined sewer overflows and storm water discharges, combined with northeast winds and wind currents, transported this material to our beaches. A large portion of that material came into the New York Bight, particularly from New York City and the Fresh Kills landfill.

In May 1987, we had a rash of plastic medical-type waste and grease balls. Grease chunks covering human waste washes up on beaches in Ocean and Monmouth County. This was originally attributed to sludge dumping, particularly the digester washout from barges. We now believe that the major cause was related to the combined sewer overflows in New York and New York/ Newark Harbor, combined with wind and current conditions that brought it south to Monmouth and Ocean Counties.

Trash and other floatable debris again hit our beaches in 1987 and 1988 with widespread press coverage. Over a billion dollars of tourism was lost due to the adverse press and the perception that New Jersey beaches were somehow dirty. Although sludge dumping was identified as one of the causes, it was not attributed as the major cause. Fresh Kills landfill was named very frequently as a possible source, as well as boats offshore. It now appears that the largest source of the floating trash debris that hit our beaches was the resuspension of floatable debris from estuary and harbor shorelines during extreme high tides and during storm water surges. Contributors also included combined sewer discharges and some sloppy practices at the Fresh Kills landfill and other areas where trash is handled along our coastline.

A final example of the problem of perception versus reality is that in the period that we have been discussing there were multiple dolphin deaths. It was reported to the press that dolphins were swimming in the sludge dumpsite and caught a disease. I do not think anybody is really sure of the full cause of those dolphin deaths.

We are aware that there is a naturally occurring bacterium infecting dolphins which compromises their immune system. That only answers half the question. If it is a natural cause why do we have this rash of dolphin deaths greater than at other times.

New Jersey is trying to do a number of things to address these problems of perception and reality. First, it is important that we realize that the public responds to visual action. When they perceive a problem, they want something done. We would like to think that what we are doing is addressing fundamental problems. It does no good to take a visible action, that is not addressing the real problem. Otherwise we are only going to have the problem resurface. We would have more garbage wash up on the beaches in a subsequent season.

Let me discuss some of the things that we are doing right now to keep New Jersey beaches clean. First, we have Operation Clean Shores. That is a program where we hire convicts to go around and clean up debris on our beaches. It starts every spring. They are picking up the litter and wood. Most of what we pick up, however, is not from beaches at all. It is in Raritan Bay, in the Arthur Kill, and in the back waters where, during periods of storm surges or high tides, material washes off the shoreline out into the ocean, and then under the right current or wind conditions, washes back onto our beaches.

In addition to the cleanup activities, we have instituted the Coastal Monitoring Program. Twice a week the water is sampled at every beach, along the bays and ocean, and monitored for coliform and other indicators. This is the most extensive coastal and beach monitoring program in the nation. It is done on a cooperative basis by the County Health Departments and the local departments.

We maintain an information network to advise the public of where problems are found. Based on this cooperative Coastal Monitoring Program we have been able to conclude that the major cause of bacterial contamination on our beaches is indeed storm water and combined sewer overflows.

This program has gotten us into some trouble. You see, New Jersey tells people when there are beach problems. With the most aggressive beach-monitoring program in the nation and with easy access to it, the press picks up reports and says we have problems at our beaches. This leads to a perception that New Jersey beaches, therefore, are somehow dirtier than other beaches. The reality, of course, is quite the opposite. We believe that our beaches and our coastal waters are, if anything, cleaner than many other resort areas.

We noted with great interest last year, when Congressman Hughes introduced a measure for a Coastal Monitoring Program similar to ours throughout the east coast in all resort areas that there was widespread resistance to instituting this sort of program. Florida led that resistance. I do not think there is any problem with Florida's water, but it is interesting that they are scared about perceptions that may imply there is something wrong.

We do a couple of other things in New Jersey that relate to keeping beaches clean. Most of you are probably aware that we fly a helicopter four days a week, and EPA flies one day a week for us, inspecting our beaches and looking for slicks offshore. We maintain contracts with various cleanup services, fishing boats, and the Coast Guard so that when slicks are spotted we can go out there and clean them up. We have teams of people on call, who can go out and clean up materials when and if they hit the beaches. The result is that every beach in New Jersey during the summer season is combed by inspectors looking for debris. You can be assured that the beaches in New Jersey will be cleaned every morning. We have some of the best beaches there are.

What we are doing right now is overcoming the perception that there is a problem. It is gratifying to see that tourism bookings are up and the press is favorable. People are recognizing the quality of the water that we have at the shore. However, we are still very vulnerable. When the *Asbury Park Press* reports that there is material washing up, for example three drums on a coastline of 112 miles, due to somebody apparently knocking them off a pier, this gets front page press. The perception that there is a problem in New Jersey continues, much to the detriment of all of us, to the Department, to our economy, to the beach users, and to the communities. And I think it is also to the detriment of scientists because that sort of statement does not represent good science. It represents a form of yellow journalism.

Let me return to the issue of sludge. I have been working with sludge policies over the last year. We recognize that when 17 March came, the problem with sludge management would not end. Right now in New Jersey roughly 55 percent of our sludge (this is after ocean dumping ends) is hauled to other states for disposal. Most of this is landfill, some of it to as far away as Texas. The Passaic Valley Sewage Commissioner has a train leave Newark everyday for Tyler, Texas, with 55 cars filled with sludge. You can imagine how that is received and perceived in Texas. Approximately 23 percent of our sludge is used beneficially either in land application or as landfill cover. Finally, about 22 percent of our sludge is incinerated.

We realized that when ocean dumping of sludge ended we could not rely forever on hauling it out of state, but there is a perceptual problem about sludge. Just the word conjures up human waste in people's minds. Never mind the fact that with secondary treatment, sludge is not human waste; it has become a processed product. It is a bacterial cell mass. It settles out after the biological treatment of sewage. Nevertheless, it is conceived of as human waste and as intrinsically evil; nobody wants it and nobody wants to let us do anything with it.

I will call your attention to some of the headlines that just occurred over the last several weeks. In the *Trentonian*, my favorite paper, "Sludge dumping; Governor Florio allows sludge dumping in the Pine Barrens." No such thing actually happened. It is true that some of the sludge is composted from Camden County Municipal Utilities Authority and is land-applied for the restoration of certain sand quarries in the Pinelands. We look at that as a good thing.

I had the opportunity last week to fly over the Pine Barrens and most of southern Jersey on a perfectly clear, beautiful day. What I saw surprised me. Throughout southern New Jersey there are vast areas of deserts, sand quarries, contaminated sites, construction sites, thousands of acres on which nothing is growing. Why not? It is because we have removed the top soil which was precariously thin to begin with. Nobody has replaced or rebuilt that top soil, and vegetation can not survive.

We have, with sludge, a product that has valuable capabilities as fertilizer, which after composting can rebuild soil structures and rebuild the top soil. I would love to be able to use that on these deserts in southern Jersey.

But I am faced with a perceptual problem–i.e. that the *Trentonian* would report that Governor Florio dumps sludge on the Pinelands. How do we overcome this perceptual problem and allow the use of sludge that has beneficial properties? It is going to take a massive effort. It will take more than simply good science. It certainly does take good science, however, to insure that what we are doing, and what we want to do, does not create new problems. It also takes public education and a concerted effort from a lot of concerned people recognizing that this is not a waste and that this is the right thing to do. When we dump sludge in the ocean, when we incinerate it, when we apply it to a landfill, we are not recovering the valuable properties of this material.

The Department is in the midst of implementing a beneficial use policy. As many of you are aware, last November the Department issued a white paper on sludge management, perhaps the most comprehensive paper on a state's sludge problems ever issued in this country. Basically, it was so comprehensive because we know more about sewage sludge than any other state.

Subsequent to the issuance of the white paper, we received many favorable comments and a lot of interest. We held a series of roundtable discussions in February and March 1991 with interested parties from all sides of the sludge management issue to develop new policies for the state. Consensus was achieved in those round-table discussions on a wide range of subjects.

The first major consensus was that we should treat sludge as a material that has beneficial properties and that can be recycled; we can recover the nutrients in the soil-enhancing value of that material. In other words, we should not dispose of it. We should focus on beneficial reuse.

A second major policy consensus was that the entire state needs a consistent high-quality sludge standard which is defined as acceptable for perpetual application on land without degrading it. That way we would not create new contaminated sites. Further, until sewer authorities achieve that quality of sludge, they must have aggressive pretreatment programs. They must examine sources other than industry that contribute to sludge contamination, e.g. homeowner practices, background concentrations in the water, and the leaching out of copper and lead from plumbing systems.

We further said that New Jersey should lead the way in beneficial reuse of sludge. We should encourage other people to use sludge beneficially on our own lands, i.e. construction sites of the Department of Transportation, as cover material over Superfund sites under our control, and application on state-owned agricultural lands. There is one additional consensus that I should mention and that is that if sludge meets this high quality for perpetual use, then we should deregulate the material and not count it as a waste. We should also actively start removing the stigma attached to this material and have it considered to be comparable to the compost that comes out of these 240 leased composting centers across the state.

We are now in the process of implementing this new policy. Commissioner Weiner in his remarks to the House Appropriation Committee on 29 May specifically endorsed these policy initiatives—initiatives agreed upon at the sludge roundtable discussions. Our timetable is such that we will be developing the high quality standards and regulatory proposals over the next several months. We will probably have them out for public review and discussion within the next several months. Our goal is to adopt regulations by the end of this year to implement the sludge management policies.

Again, let me point out another perception and a small advertisement for the Department. There has been a perception that the Department is a problem agency, with criticism of permit delays and lack of direction. I would like to make everybody aware that there is a lot going on in the Department of Environmental Protection. In the six-month period between May and October 1991, there will be more regulations and more standards in the water area by the Department than in its history or the history of its predecessor, the Division of Water Resources.

Every one of these regulations will be presented in a public forum. We will hold public meetings and solicit input from the public; we need the input of good scientists in New Jersey and throughout the region as well.

The regulations will include surface water standards, particularly the adoption of new quality standards for all surface waters including the ocean and estuary. They will include toxic contaminants. We will be issuing or reissuing ground water standards for the entire state. We will be issuing standards in policies related to practical quantifiable levels. Practical quantifiable levels (PQL's), are those levels of toxic contaminants that can be reliably and accurately measured. They are differentiated from method detection limits, which are lower than PQL's. Method detection limits indicate that we can detect it, but we can not reliably quantify it.

PQL's are particularly important for application of surface water and ground water standards and the regulation of discharges. We need to adopt policies on processes when the surface water or ground water standard is below the PQL. How you regulate discharges in that case is an interesting policy to debate, and certainly needs the input from the scientific community.

We now have cleanup standards, for the first time—uniform, consistent cleanup standards for the entire state for contaminated sites. They address both how clean we expect the soil to get and how clean we expect the ground water to get, and the thorny problem of what happens if you can not attain them. In some cases, you could pump and treat forever and never achieve the cleanup standard. How do you address that? What do we do about areas where there is existing widespread contamination within the state? These are addressed in the cleanup standards, and there is a series of four public meetings which are ongoing in this topic area.

The Clean Water Enforcement Act, which was signed by Governor Florio last year, is coming into effect on 1 July 1991. Regulations that were proposed in April for enforcement of this Act will be adopted in July. Further regulations coming up to implement the Act are in the area of pretreatment and delegation of pretreatment programs to sewer authorities.

In September, we expect to have an extensive regulatory change reforming our New Jersey Pollution Discharge Elimination System (NJPDS) permit system; cleaning it up and making it more reasonable, which will expedite the process while at the same time protecting environmental quality.

We are now issuing combined sewer mapping standards. We are issuing new regulations on Discharge Pollution Control and Containment (DPCC) Plans. This is in response to the Oil Spill Prevention Act signed last July. They are one of the most aggressive sets of regulations in the country, designed to prevent discharges of hazardous substances and contaminants both from stationary facilities and marine transfer activities.

We will be proposing sewer ban and treatment work approval reform regulations, a frequently criticized program. When we review it, the perception of what good this program is doing may mesh with the reality of programmatic good.

That is quite a list of regulations, all of which affect water quality. Taken collectively, this set of reforms will basically overhaul the entire system of regulation in New Jersey. This is a critical juncture for regulation of water discharges and regulation of water quality throughout our state. The Department is doing this in the midst of a severe state budget crisis, hiring freezes, and the perception that we are somehow an embattled Department. I hope that some of what I have said has put that perception to rest.

To return to my theme about perceptions versus reality, we need the help of the scientific community to tell us what is reality. What are the problems on which we should be focusing? What will be the impact of regulatory programs that the state is proposing?

We also need the help of a broader community to change perceptions. And I think that when you consider results of your research, you should spend time thinking how to get the message not only to the scientific community but to the regulators and public both in terms of how we regulate the environment and the policies we adopt. Do not let the journalists and politicians control the day.

Historical Background of the 12-Mile Dumpsite

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In the mid 80's and then in the late 80's, I headed teams to deal with the statutory deadlines for getting sludge moved from the 12-mile site and then out of the ocean. Consequently, I have looked at this symposium talk as a labor of love and have tried to come up with some historical facts that I thought might be interesting.

The last barge went to the 12-mile site on 31 December 1987. This ended over 50 years of use that started in 1924. John Keith has described what sludge is; the only thing I would add is to remind you that it is around 96 percent water, i.e. the volume is predominantly water, not solids. In 1884, New York City built their first sewage treatment plant. It was the first in the area, and by the early 1900's it was a growing business. So much so that by 1927, New York City had built the world's largest [plant] at Jamaica Bay.

The first ocean dumping in this area actually occurred at a local site in 1917 but it was not sewage sludge. There were no Federal regulations within the Bight and in Coney Island Sound, and dumping consisted of street sweepings, garbage, and refuse largely from New York City. They went out and dumped it, and the winds blew in the wrong direction. You can guess what happened. There were washups on the Long Island beaches and the City was taken to court. In 1931, the U.S. Supreme Court ordered the dumping of garbage and street sweepings stopped. The last barge went out on 28 June 1934, some three years after the statute.

While all this was happening with the garbage, the metropolitan area of New Jersey was developing as well. The area covered by the Passaic Valley Sewage Commission had a population of around a half a million people, and sewage disposal was becoming a problem. Sometime around 1910, Passaic Valley came up with the proposal that they wanted to discharge sewage into the inner harbor. Obviously, New York City's sanitation district, the Health Department in those days, became upset. So they went to the Supreme Court to stop New Jersey. Although there were attempts throughout that period by both states to arrive at a mutual solution most of these efforts were futile.

The Supreme Court came out with a decision in 1921 with regard to this court suit on the Passaic Valley, which basically said, "Guys, go solve your problem; we have got better things to do; try to cooperate." So the states started to cooperate, and as a consequence a site was designated for sludge dumping.

The sewage sludge site was in approximately 90 feet of water, 6.6 square nautical miles in area, 10 miles east of Highlands, New Jersey, and 9.9 nautical miles south of Long Island. Obviously, the criteria for its location was to insure that it was outside of the navigation lanes and in an area where the beaches would be protected from sludge materials washing back on to the shore.

Meanwhile business was still growing. By 1927, New York City had 11 so-called "filtration plants." Interestingly in 1927, much of that sludge was not dumped in the ocean. As late as the early 1940's for example, Coney Island sludge was taken to Marine Park and put in shallow lagoons where it dried and was then used as humus on park lands in the City of New York. The idea of land-based use of sludge is not really something new; certainly not new in this area, and I think the rest of the country has been going in that direction a long time.

The product unfortunately grew. It grew in size and complexity and simpler solutions became a desired path. In 1937, an estimated 11 dry tons of sludge was produced in the city of New York. By 1947 that number had risen to 300 dry tons. During World War II fat was skimmed off the sludge and recycled, and sold at 80 cents per hundred pounds. Whether they used the fats for lubrication or soap, I am not sure.

At peak use, there were almost 200 sewage treatment plants discharging at the 12-mile site. However, by December 1981, there were only nine: in New Jersey, Bergen County, which jointly treated sludge from Essex and Union Counties, Linden, Roselle, Middlesex County, Passaic Valley Sewage Commissioners, and

Rahway Valley; in New York, Nassau County, Westchester County, and New York City.

Let us review the more recent history concerning this site. Under the 1972 Marine Protection Research and Sanctuaries Act (MPRSA), the site was designated as an interim disposal site in 1973. That basically meant we did not have an Environmental Impact Statement for the site, but since it had been used historically, we allowed its continued use. In 1973, we started issuing the first of our permits for dumping. These were issued through 1976 with cessation required by the end of 1981. Again, EPA had taken a rather strict interpretation of the Act as far as what was and was not harmful. Basically the Agency was saying that unless you could prove there were no other alternatives, you had to get out of the ocean. We started having hearings on the designation of the site as well, including an Environmental Impact Statement. History began repeating itself in 1976.

Some interesting things happened in 1976. One, there was a series of pier fires; I think three piers burned down at the beginning of the year. Two, there was a massive explosion. Some kids had decided to see what it would sound like if they dropped a firecracker into a sludge holding tank in Bay Park, Long Island, and two of the tanks blew up, spilling a million gallons of sludge into the bay. These events coupled with a lot of rain and sustained southerly winds tended to drive everything back on shore and the politics of the situation started taking over.

Congress was quick to act. We already had issued permits that required cessation by 31 December 1981. That not withstanding, in November of 1977, under President Carter, Congress reaffirmed that 31 December 1981, was the date to close the site. However, there was no clarification as to what was reasonable and unreasonable degradation. The Environmental Protection Agency (EPA) had to balance the environmental impact of sludge dumping versus the environmental impact of land-based solutions. One of the reasons this conference is so interesting is to consider how these findings relate to all the monitoring that is now conducted at the 106-mile site. When you start debating the impacts on land, they are easy to grasp; you can figure out the impact, whether it is an air impact from an incinerator or other source. It is relatively simple and usually relatively localized. The problem with dumping in the ocean is that you can not figure out impacts. You think you know what they might be, but you are never really sure. You keep saying it might be this, or it might be that, and people ask "Well, is it definitely?," and you can not answer. The old quote is that people do not live in the ocean, so the constituency in the past has been rather obscure. In 1976 the politics were such, however, that the statute was changed.

In May 1979, we approved the 12-mile site as a legal site. At this stage, the impact statement was completed and we scheduled that use of the site would expire on 31 December 1981. We continued in earnest to provide grant as well as regulatory assistance to municipalities to get the nine dumpers out by 1981.

On what did we base the decision not to redesignate the site? There were shellfish bed closures because of bacterial problems. There were elevated heavy metals and toxic organic compounds in the bottom sediments. There was reduced dissolved oxygen in the area and there were reported changes in diversity and abundance of the marine life. We used these effects as our key reasons not to redesignate the site.

As mentioned earlier, in 1981 New York City and a couple of other municipalities took us to court and we lost. The decision was that EPA had no right to presume unreasonable degradation. It barred us from initiating any action until we looked at the whole issue and basically required us to do a balancing act.

In December 1982, EPA agreed to reopen the site and keep the process going, requesting comments on the redesignation. In May 1984, we announced our tentative decision to deny the redesignation and started the process of designating the Deep Water Municipal Sludge Dumpsite which had been previously called the 106-mile site. The designation recognizes other ocean sites in that area where industrial dumping occurred. In April 1985, we formally denied the existing permits and established a phase-out schedule by 31 December 1987. On November 1986, Congress took what we already had in the consent orders as a basis and adopted it as a requirement of the law.

One of the problems, to give you a little insight into negotiations, was that to get to and from the 12-mile site and dump a load of sludge was a relatively simple event. Weather problems were not great; the distance obviously was not great. You could get a barge back and forth in one day with no problem whatsoever, nor did you need an ocean-going barge. Moving to the 106-mile site required larger ocean-certified barges, and the passage of time to and from the site took about three days. We had to start looking at barge capacity and to bring in new barges.

New York City at this point is the only sludge dumper and, for the last several years, has been the only one that owns its own fleet. Everybody else dumps by contractor. New York City used self-propelled vessels, about 140 feet long—nothing too large. Those vessels are no longer used to go out but are used for lightening operations within the City. The replacement barges are huge—more than 300 feet long. Vessel acquisition became a problem and was a factor in the negotiations.

From 1973 to 1987, it is estimated around 90 million wet tons of sludge were dumped at the 12-mile site.

That interval was the only time it was dumped via a Federal permit. Before that, remember, it was dumped without a Federal permit because there was no Federal requirement.

The site was finally de-designated on 2 February 1990 and no longer exists as a site. Of the dumping that went on within the apex and within the Bight as well, all the sites except for the one designated for dredge spoil have been or will be closed shortly. Even the sludge dumpsite at the 106-mile site actually expires on 31 December 1991. New York City will dump a bit longer. They will be dumping under court order. If they go into violation of that court order the site could theoretically disappear on them as well. That was a strategy that EPA used with the Department of Justice in negotiating the agreements. Now you are up to date on the 12-mile site.

On the 106-mile site, you can watch political effects—in 1987 from the New Jersey washups and in 1988 from the Long Island washups. On the Long Island washups, I received a frantic call on 6 July 1988 that medical waste was washing up bountifully on the shoreline.

The estimate was that they had bags of medical waste. I said, "Do not do anything, I will be right there." (I happen to live in Nassau County in the town of Hempstead.) A helicopter picked me up and I was out there in an hour, walking the beach. Most of what was reported as medical waste were firecrackers, the tops of rockets from the 4th of July. The medical waste from a 10-mile stretch of beach might have filled one baggie. It was a very slight amount of medical waste. But between what happened in New Jersey in '87 and what happened on Long Island in '88, this was the fatal blow to sludge dumping. Our estimate is that basically none of it came from sludge. It comes from combined sewer overflows (CSO) for the most part. To bring you up to date, in the spring of 1989, EPA brought together a joint task force of the states, EPA, the Coast Guard, NOAA, and the Corps of Engineers, and said between all of us we had some resources. I guess some of us had some idea at the time of coping with a resuspension problem at high tides and a combined sewer overflow problem. We developed something with the Corps of Engineers that had two vessels out there, actually three, where they would use those vessels to buy new nets with a smaller mesh and basically skim the harbor. Through a network, with EPA as central command, EPA and DEP helicopters would basically fly the harbor shorelines, identify slicks, and direct Corps of Engineers vessels to skim those slicks. New York City agreed to allow us to take that waste and put it in the Fresh Kills landfill. And it has worked amazingly well but does not solve the problems. We have provided to the City of New York funds for two of these vessels at a cost of roughly six million dollars. These ships will operate on the Sandy Hook/Rockaway transect. New York City will provide for their operation and maintenance on a regular basis.

So, that coupled with storm water strategies and CSO strategies are being put in effect as well with the states. We hope to solve the floatables problem. The sludge cessation really has all been related to the scare and perception that floatables on the shoreline have been brought on by sludge dumping. It is not true and the advantage of the situation is such that we have gotten the sludge out of the ocean.

It has been a long and torturous history for some of us, and I am glad to see it coming to an end.

Audience Questions

Question: Richard, I notice that neither you nor John [Keith], mentioned George Whidden and the Coalition to Cease Ocean Dumping. Any comments on the role that played in public perception?

R. Caspe: There are a lot of people saying a lot of things. It played a role; there is no question about that. A lot of people had the perception that terrible things were happening as a result of sludge dumping. The arguments on why sludge dumping should end were not really based on technical [facts]. The public policy view of what should and what should not be done with sludge probably came out the right way from my perspective. The end result was good, but if you listened to how the decision came about, there were a lot of things being said by both sides. I heard people from New York City testify that sludge was a valuable resource to the area; that without that there would not be any fish in the area. They considered it a nutrient-starved regime without sludge. There were different people saying different things.

Experimental Design of the 12-Mile Dumpsite Study

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ABSTRACT

The study of effects of cessation of sewage sludge dumping at the 12-mile site used a sample design based upon the technique of replication in time. This design yields unbiased estimates of the ecological effects of sewage sludge by precluding the confounding effects innate in many environmental impact studies. Three sites were selected for intensive subsampling based upon an assumption that they represented a gradient (maximum, intermediate, minimum) of exposure to sewage sludge. In odd-numbered months plus August, eight complete subsamples were taken at each of the sites (the Replicate Study, 576 samples). A complete sample consisted of a bottom-trip hydrocast, a benthic grab, and an otter trawl. In even-numbered months, single complete samples were collected at 22 sites distributed about the dumpsite area (the Broadscale Study, 415 samples). This part of the study was conducted to allow extrapolation of the experimental results to a wider area. The experimental design was developed to maximize information in the context of the constraints and caveats of both the statistical model and the availability of experimental resources. The essence of this particular design is that the likelihood of the results being biased (by random effects or by the confounding due to nonrandomized sampling) is minimized, thereby yielding useful management information.

Introduction

Previous studies of the fate and effects of sewage sludge disposal on the benthic ecosystem of the New York Bight have relied on correlative techniques to determine relationships between sediment characteristics and infaunal community structure (Pearce et al., 1981; Reid et al., 1982; Reid et al., 1987). Although the demonstrated relationships between chemical/physical properties and indicator species assemblages are likely to be true, the strength of these relationships with respect to disposal cannot be determined quantitatively. When the structure of a study lies outside the paradigm of an experiment, it cannot be assumed that all the possible variables or cofactors which may confound an analysis have been measured or taken into account. For example, assume samples are post-stratified such that contaminated depositional sites are compared statistically to uncontaminated depositional sites. There are unresolvable innate differences between the treatment groups. The distribution of disposal materials is contiguous; thus the groups are likely to engender different base communities or differential exposure to other sources of contamination. Such a comparison is confounded by location. The only way to quantify the effects of disposal is through experimentation.

Although it is impractical to conduct a classical field experiment on the effects of sewage sludge on the benthic ecosystem of the New York Bight, the planned closure of the 12-mile dumpsite afforded the Environmental Processes Division (EPD) an opportunity to design an experiment using the method of replication in time (RIT)[1]. A general description of RIT is given by Stewart-Oaten et al. (1986); a detailed statistical description of the application to the 12-mile dumpsite study is given by Pikanowski (unpub. data). Besides the usual resource constraints, there are constraints on the sample design imposed by the method. These constraints or caveats are explicated in Pikanowski (unpub.

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[1] Not all of the individual sub-studies are included in this design strategy.

data) and are presented here without justification. Although the theory behind RIT may be somewhat complex, the execution is quite straightforward.

A contaminated site and an uncontaminated site are selected. At intervals, before and after termination of disposal, samples are collected simultaneously at the selected sites. The difference between the values measured at each site is calculated, yielding a replicate in time. Assuming no confounding effects are operant (Pikanowski, unpub. data), a comparison of the "before-closure" and "after-closure" replicates yields an estimate of the effect of termination of disposal. If an ecologically significant difference is specified a priori, the comparison yields a test of the null hypothesis that there is no effect of termination on a particular response variable. In fact, the intensity of disposal declined from 1.72 million wet tons in the third quarter of 1986 (the first samples were collected in July 1986) to 0.69 million wet tons in the fourth quarter of 1987, prior to cessation of all dumping in December 1987. Thus the treatment was not "switched off" but rather stepped down. Likewise, there is a possible lag in "recovery" with respect to the end of disposal. Linear trends due to this variable treatment intensity may be detectable by tests for serial correlation (see below).

Two qualitative judgments must be made as to the validity of the results. First, a variable that evidences no significant change may have been altered irreversibly by the introduction of sludge. Because the treatment materials (nutrients, contaminants, pathogens) have limited lifetimes and the geographical extent of the population pool of New York Bight species is much larger than the affected area, this possibility is unlikely in the long run. However, it is possible that some effects (for example, recruitment of long lived benthic species such as the surf clam, *Spisula solidissima*) may not be reversed during the short-term course of the study. Second, the response of the ecosystem to an onset of disposal is not necessarily the inverse of the response to cessation (particularly in the short term). Whether it is valid to extrapolate the results to an onset of disposal must be judged qualitatively.

Site Selection

The first step in the design process is site selection. Details of site characteristics are given in the Plan for Study (EPD, 1988). Based upon the historical studies previously cited, selection of the contaminated site was obvious. This site (NY6, Fig. 1) was found to be maximally contaminated; sludge accumulation was sometimes manifested by an observable layer of flocculent material. NY6 is near the northwest corner of the sludge dumpsite (Fig. 1); this area received a large percentage of released material because it was proximal to the ports of origin. Net transport of sludge followed the gravitational gradient west into the Christiaensen Basin, thus accumulating at NY6.

Selection of the uncontaminated, or reference, site was not as straightforward. It is implicit in RIT that the reference site will respond similarly to common large global events (in the sense of affecting the whole study area) unrelated to the treatment. Storms, climate change, or recruitment success are examples of such events. There is no certainty, beforehand, that a reference site will be a good one in this respect. Thus, two sites were selected as possible reference sites. Station NY11 (Fig. 1) is located south of NY6 on the eastern shoulder of the Christiaensen Basin and showed little evidence of sludge contamination. Station R2 (Fig. 1) is north of NY6 and evidenced an "enriched" benthic community with high biomass of several tolerant, though not necessarily pollution-indicator, species. It was not clear if this enrichment had a dumpsite or estuarine origin (or a combination of both). Station R2 served a dual purpose; it was either an alternative reference site or represented a treatment site at an intermediate level of overall contamination (derived from disposal and estuarine sources or both). The three stations were similar in depth and underlying sediment type (fine sand), and they were unlikely to be affected significantly by dredged spoil disposal (Fig. 1) on the western side of the Christiaensen Basin.

One of the caveats of RIT is that results apply specifically to the contaminated site (NY6). This site repre-

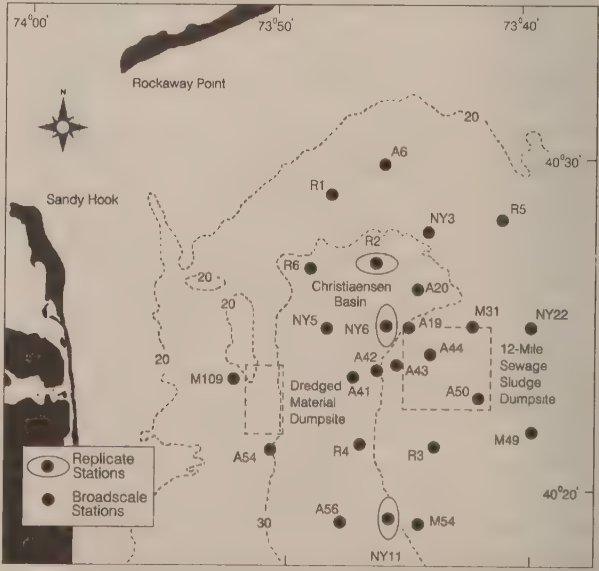

Figure 1
Locations of stations sampled on the replicate and broadscale surveys during the 12-mile dumpsite study.

sents only a small portion of the affected area; an area-wide extrapolation of the point estimate at NY6 requires additional information. To this end, a suite of broadscale stations was selected for study (Fig. 1). These sites were chosen to provide a comprehensive overview (geographically and in terms of treatment gradients) of the area of the inner New York Bight potentially affected by disposal (EPD, 1988).

Sample Allocation

Preliminary cruises indicated that four complete samples could be reasonably collected per cruise day. A complete sample consisted of a bottom water sample, a benthic grab, and an otter trawl. A complete list of variables and a description of sampling protocols is given in the Plan for Study (EPD, 1988). All variables were measured synoptically for two reasons: first, synoptic collection would provide a standard, efficient correlative data base in the event that the experimental aspect of the design failed; second, it is difficult to optimize sample allocation at sea. Logistic constraints would allow at least six days of sampling per month. Thus, 24 complete samples could be collected reliably per month. The study was scheduled to begin in July 1986 with the dumpsite scheduled to close at the end of December 1987; thus, 18 months would be available for sampling in the before-closure period. Sampling was scheduled to continue through September 1989; thus, 21 months would be available in the after-closure period. The design problem was to maximize the information generated under these constraints.

Ideally, one would like to sample for a much longer time, both to increase the chance that responses within the sampling period would be representative of the long-time average and also to allow the system to equilibrate to the absence of sludge inputs. Because this option was not available, sampling needed to be intensive. For reasons developed by Pikanowski (unpub. data), monthly samples at a site were averaged; the difference between these site averages yielded a monthly replicate in time. Although statistically less efficient than calculating a replicate for each sample pair, the loss in efficiency is minimized for measures that evidence high sampling variability within sites (for example, abundances of finfish or macroinvertebrate infauna). Thus, for stations NY6, R2, and NY11, eight subsamples could be collected per month per site, yielding one replicate per month for each sample pair (NY6-NY11 and NY6-R2, the Replicate Study).

If the effect of disposal at NY6 were the sole interest, 18 before-closure and 21 after-closure replicates could be generated. However, a measure of the area-wide response is useful in calculating the total environmental costs. Thus, it was decided to sample 21 stations once per even-numbered month (the Broadscale Study). The three replicate sites would be sampled in odd-numbered months plus August. The doubling of effort in August should be worthwhile; low dissolved oxygen concentration in late summer causes the reducing sediment layer to rise toward the surface. This may contribute to dramatic changes in the infaunal community structure and possibly affect fish and epibenthic megainvertebrate distributions as well.

A trade-off was made between the power of the study to resolve changes at NY6 (a reduction in total replicates from 39 to 24) and the ability to extrapolate results (via correlative techniques) in an informed manner to a broader area. It remains to be seen whether this was a wise choice. It should be noted that, for variables that have a small sampling variance or evidence a large response to termination of disposal, the method of RIT can be usefully applied to the Broadscale stations.

Allocation Details

Sampling at the same place eight times within two weeks is likely to lead to nonindependence between samples due to removal of both sediment and biomass (sampling without replacement). Thus, it was desirable to designate a site more extended than the footprint of the sampling devices (trawl and benthic grab). The extended replicate sites that were chosen are represented by ellipses (Fig. 1). These ellipses are large enough to accommodate one km trawl tows and are oriented with the local bathymetry to minimize variability in depth. Extending a site has the drawback of increasing within-site variance. A design compromise was struck between these conflicting practicalities. Three complete samples would be taken at the center of the site (designated as "radial" samples). Certain sediment chemistry and bacteriological measurements were collected only in these radial samples. The remaining five samples would be taken at points on the perimeter of the ellipse (designated as "distal" samples). Initially, the five points were selected at random, within the constraint of being uniformly distributed about the perimeter. Modifications to the initial site selection were made to allow unhindered trawling. These points were revisited on subsequent sampling occasions to reduce within-site variability and ensure unhindered trawling.

For variables with highly heterogeneous distributions within sites, significant responses may best be detected by using only the three radial samples. For those measures that are not very heterogeneous, or are so heterogeneous that the radial samples are as dissimilar to each other as they are to the distal samples, using all eight samples should be most efficient. This partitioning also allows an ancillary study of the scale of heterogeneity for various environmental measures.

The RIT method assumes that samples are collected simultaneously at both the contaminated and uncontaminated sites. Obviously, this cannot be done from a single sampling platform. In fact, the time constraint of day trips allowed for two samples to be collected at a site in the morning and two samples collected at another site in the afternoon. Some variables (for example, stomach contents or distributions of fish and megainvertebrates) are likely to vary within and between days. This diel and daily variability cannot be removed; however, it can be made unbiased by blocking out the effects of morning/afternoon, first week/second week, and site-pair sampled as shown in Table 1.

Statistical Methods

Whether the before-closure and after-closure means differ significantly can be tested simply with either the Student's-*t* or Mann-Whitney *U* tests (Zar, 1984). In most cases, the tests will give the same result. Contradictory results may occur if borderline cases are otherwise suspect or if the data are severely non-normal. Unfortunately, there are too few replicates to test reliably for normality, thus the choice between parametric and nonparametric testing is not deterministic. Regardless, ecological and statistical significance may not be equivalent. It is possible for an ecologically important difference to be declared statistically nonsignificant because the standard error of the difference between means is large compared to the treatment effect. Conversely, trivial ecological differences can be declared statistically significant when the standard error of the difference between means is relatively small. It may be more useful to generate confidence intervals about the difference between means (parameter estimation rather than hypothesis testing) or, for a manager, to post-calculate the power of the experiment to detect a difference that may be deemed ecologically or socially significant (Pikanowski, 1992). In the latter case, the manager can then decide whether the experiment was successful and the results are acceptable.

Table 1

Sample schedule used in the 12-mile dumpsite study (replicate survey) to block out effects of morning/afternoon, first week/second week, and site-pair sampled (modified from EPD, 1988).

Sample day	Week	Morning site	Afternoon site
1	1	NY11	NY6
2	1	NY6	R2
3	1	R2	NY11
4	2	NY11	R2
5	2	NY6	NY11
6	2	R2	NY6

The results of RIT can be confounded if there is significant serial correlation between replicates within a period (before or after closure). This can occur if the treatment level changed during the study or if a large event caused an effect (such as differential recruitment of an infaunal species) at one site only and this perturbation carried over to succeeding replicates. An event of this kind may well bias a period mean. By using the c or runs test as given by Zar (1984), serial correlation may be detectable. For some variables, serial correlation may be expected in the after-closure period because the system needs time to equilibrate. In the presence of this type of correlation, the real difference between period means is likely to be larger than the experiment indicates. For some variables (for example, total organic carbon), there may be an approximate linear trend in the before-closure period corresponding with the stepped phasingout of disposal. Removing this trend by factoring in the disposal rate will decrease the variance estimate and will improve the precision of the test. If serial correlation is detected in the before-closure period but cannot be explained by a covariate, results should be interpreted guardedly; it is not reasonable to predict the course of trends unrelated to the treatment.

Some variables may be seasonally nonadditive; that is, the difference between sites may be correlated to the magnitude of the site values. The results will not be biased if the before and after periods cover the same calendar intervals; however, the increase in variance due to seasonal nonadditivity will diminish the precision of the test. The 12-mile dumpsite study time frame does not encompass similar calendar intervals in the before and after periods. The study time span is too short to allow filtering of seasonality; thus, if this effect is detectable (Pikanowski, unpub. data), a reduced data set encompassing similar calendar intervals must be used. If a species occurrence is highly seasonal, the number of nonzero data points may be insufficient to allow for a test (Pikanowski, 1992).

Conclusions

Correlative studies and post-stratified environmental impact designs are invariably confounded. That is, quantitative estimates of an environmental impact are invariably biased and are thus unreliable. The essence of the replication in time design is that it is an experiment; the treatment is replicated over virtual experimental units. These virtual experimental units are not "true" experimental units—they are the same micro-environments sampled at different times. However, the possibility of confounding events is minimized both by the reference site, which eliminates global random effects, and by testing for serial correlation, which guards against local random effects. The results are quantitative estimates of the effect of the environmental impact that are not likely to be confounded (when serial correlation is not detected). Thus, reliable management information is generated.

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Audience Questions

Question: Obviously, substrate life or ocean floor habitat life has a great bearing on the structure and the species of life. How do you wrestle with that factor in dealing with control versus experimental sites where you have got 10 or 20 feet of depth difference? Have you looked at that aspect of the problem?

R. Pikanowski: We have selected our sites so that the depths would not be that different. You are right; I think there may be a ten foot difference between one of the controls and the impact station. The key to this is that the stations do not have to be exactly alike, just that they respond in a similar manner. In other words, if there is a natural increase in abundance of a particular species, it will settle in the two sites somewhat equally. This may not happen for everything. Some of our data indicate certain species, such as *Capitella,* are not found anywhere but at the impacted station. So you do not even need a control in that case; you can just compare the before and after data. No two pieces of the environment, no matter how close they are, whether they are land or sea, are ever exactly alike. And that is one of the problems you have in doing environmental experiments. You cannot make the assumption that the things will be alike–we are making an easier assumption, that they respond alike.

If you examine the plot of bottom temperature at our three study stations, the interesting feature is that the summer of 1988 was very cold. Now, had we done a simple before and after study, just comparing NY6 or some other contaminated station, our results for variables influenced by temperature would have been confounded by this cold summer. If you examine a plot of the replicate stations and examine the difference between the control and impact areas, you see that effects of the cold summer of 1988 really do not appear anywhere. The hope is that whatever its influence was on the ecology and the variables that we have measured, it had somewhat the same impact at both our control and contaminated stations. However, it may not have the same impact, and that gets into the nuts and bolts of statistics. We have ways of testing for that to some degree.

Question: I have a question about the location of the three replicate stations. It seems they are a bit west of the sludge disposal site and I wonder if it might be affected by anything that might have happened at the dredged material disposal site? Why were they west of the 12-mile site instead of right on top of them?

R. Pikanowski: From historical studies we found that sludge did not accumulate very much at the actual dumpsite. It is not hard to imagine that by following the bathometric gradient—it settled into the Christiaensen Basin. We found that NY6 was at the focus of where the sludge resided. NY11 and R2 are both on the other side of the Christiaensen Basin from the dredged disposal area and we hoped that this would be somewhat of a barrier to dredge effects.

Question: If that was your hope, do you know whether the dredged material disposal site is still being used?

R. Pikanowski: Yes.

Question: What kind of criteria do you have to determine whether you have a clean site?

R. Pikanowski: We used all the data developed during the MESA program. It may be possible that somebody will tell us that NY11 was not as clean as we thought.

Question: Could you say a few words about the platform that was used for the sampling?

R. Pikanowski: Our sampling platform is the now-decommissioned research vessel R/V *Kyma.* The crew, Fritz Farwell, Captain, and Sherman Kingsley, Mate, made this study far less of a horror show than it could have been. I once ran a listing of the Loran readings for our broadscale stations and I noticed that they were almost identical. One month, we got fairly large deviations and then I realized that that was the month Fritz did not sail the boat: he is extraordinary. He and Sherman Kingsley made life bearable.

One reason that we called the eight monthly samples at each station "subsamples" and "replicates," which has some statistical implications, is because we had the one platform. We really could not sample things simultaneously, and there was a certain amount of lag time between collecting the samples at one station and then cruising and starting at another. With time lag and tide changes, simultaneity goes out the window. So, we did block the experiment with week 1, week 2, morning, and afternoon, which is described in the Plan for Study [EPD, 1988].

Introduction

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This first set of four papers addresses just one hypothesis, which states

> "The cleansing of sewage sludge from the Christiaensen Basin will be accomplished by episodic down-valley transport of sludge which can be related to windfield conditions."

The state of our knowledge concerning factors related to this hypothesis at the outset of the study was as follows.

Bathymetry

The Christiaensen Basin, located at the head of the Hudson Shelf Valley and defined by the 30-m contour, served as a sink for a large portion of the sewage sludge dumped in the 12-mile dumpsite (12-MDS) immediately to the southeast of the Basin. It was unknown at that time whether the sink was permanent, but it was suspected that the Basin was serving only as a temporary sludge sink. A 1982 study of the concentrations of *Clostridium perfringens* spores found levels on a transect from the Basin along the axis of the Hudson Shelf Valley to be higher than levels on the shelf and showed that spore concentrations decreased exponentially with increasing distance down the valley to the shelf edge, about 140 km (Cabelli and Pedersen, 1982). This finding clearly indicated that sewage sludge components were being transported down the Hudson Shelf Valley, but it did not indicate the proportion of the sludge material moving along that path or the rates of movement.

Hydrography

The water mass in the inner New York Bight is strongly vertically stratified in summer and early fall but only weakly so in late fall–spring. In winter, the water column is nearly isothermal and isopycnal, and there is an onshore-offshore temperature gradient, with the coldest temperatures found nearshore. At all times of the year, the water column is susceptible to rapid changes induced by meteorological factors: wind-driven upwelling, downwelling or mixing, and seasonal increases of run-off, producing a low-salinity surface layer.

Bottom Currents

Average currents in the inner New York Bight are westward off Long Island and southward off New Jersey for both the surface and bottom. The mean bottom current speed is about 0.5 cm/s with a standard deviation of ±15 cm/s. In the Hudson Shelf Valley, average bottom currents were found to be 2–5 cm/s upvalley, but reached 20–50 cm/s either up- or down-valley during periods of strong winds, with strong up-valley flows related to strong, persistent west winds.

Sediment Transport

Little was known about sediment transport, either measured, inferred, or computed. Estimates indicated that shelf sediments at 30 m depth could be transported about 30% of the time by currents and 8% of the time by wave action (McClennen, 1973).

This base of information was the springboard from which four studies were launched into hydrography, bottom currents, sediment deposition, and sediment resuspension and transport.

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Oceanographic Conditions in the Inner New York Bight During the 12-Mile Dumpsite Study

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ABSTRACT

Oceanographic observations made in support of the 12-mile dumpsite study are used to characterize the spatial and temporal variability of water properties in the inner New York Bight. Comparisons with other long-term data sets indicate that the oceanographic conditions in the inner New York Bight during the period of the 12-mile dumpsite study were representative of the region, except during the summer and fall of 1988 when bottom water temperatures were 2–4°C cooler than normal. The summertime near-bottom oxygen gradient decreased during the study, suggesting a reduction in the seabed oxygen demand.

Introduction

An interdisciplinary study was conducted from 1986 to 1989 in order to document the response of the habitat and biota in the inner New York Bight after the cessation of sludge dumping at the 12-mile dumpsite (12-MDS) in 1987 (Fig. 1). Oceanographic data were collected during three separate sampling programs in support of the 12-MDS study. First, seven oceanographic survey cruises were conducted between May 1987 and August 1989 to collect temperature and salinity data through the water column in the inner New York Bight. Second, near-bottom temperature and dissolved oxygen measurements were made as part of the monthly sampling at the three replicate stations (NY6, NY11, and R2) (see Pikanowski, 1995, for a description of the replicate sampling design and the location of the replicate stations). Third, time series of temperature one meter above the bottom were recorded by current meters deployed during the study (Manning, 1995). With these data, the temporal and spatial variability of the water properties in the inner New York Bight are described to assist in the analysis and interpretation of the physical, chemical, and biological processes investigated in the 12-MDS study.

Hydrographic properties of the waters in the New York Bight and their annual cycle have been well described in the literature (e.g., Bowman and Wunderlich, 1977; Sherman et al., 1988; EPD [Environmental Processes Division], 1989). Generally, the water column is vertically uniform in winter because of surface cooling and wind mixing. Beginning in the spring, surface warming results in thermal stratification which intensifies through the summer. In fall, with the onset of cooling and increased winds, the stratification eventually breaks down and the water column becomes vertically uniform again. The distribution of salinity in the New York Bight is dominated by outflow from the Hudson-Raritan estuary. With peak outflow in the spring, a plume of low-salinity water extends southward from the estuary along

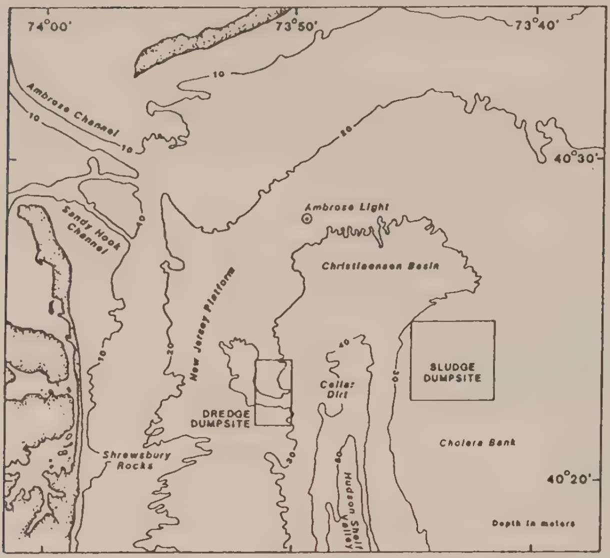

Figure 1
Location of the 12-mile sewage sludge dumpsite in the inner New York Bight.

the coast of northern New Jersey. The spatial extent of this plume and the salinity contrast between waters in the plume and the rest of the Bight is dependent upon the amount of runoff passing through the estuary.

The Northeast Fisheries Science Center, Marine Resources Monitoring, Assessment, and Prediction (MARMAP) program, collected temperature and salinity data across the northeast continental shelf for the period 1977–1987. These data provide a basis for comparing the water properties during the 12-MDS study with long-term conditions. The MARMAP observations in the New York Bight have been described by Manning (1991) and Mountain (1991).

Data and Methods

Survey Cruises

The sampling activities conducted on each of seven survey cruises are summarized in Table 1. On three of the cruises (ALB87-02, ALB88-03, and ALB88-10) the primary objective was to deploy or recover current meter moorings for the 12-MDS, and only a few transects of hydrographic stations were occupied.

The cruises in May 1987 and in June 1989 used an identical grid of stations to survey the water properties in the inner New York Bight. Two distinct surveys of the area were completed on each cruise. The cruises in August 1987 and August 1989 used a different grid of stations. Due to requirements of other sampling programs on the August cruises, the entire grid of stations was not completed before portions of it were resampled. Although many stations were occupied more than once, the sampling did not result in distinct surveys, separate in time. The first occupation of each station was used to form a single survey of the region for each August cruise.

Two different Conductivity, Temperature, and Depth (CTD) profiling instrument systems were used on the survey cruises. In 1987 and 1988, a Neil Brown Instrument Systems Mark III instrument with a rosette water sampler was used. In 1989 an internally recording Sea-Bird Electronics, Inc., SBE9 instrument was used. The procedures used to process the data from both CTD systems are described in Taylor and Mountain (1991).

Replicate Stations

As part of the sampling protocol for the replicate stations R2, NY6, and NY11, three bottom temperature measurements were made each month at the center of

Table 1
Summary of oceanographic measurements collected during the 12-mile dumpsite study survey cruises.

Vessel	Cruise	Sampling Dates	No. Stations	Coverage
ALBATROSS IV	ALB87-02	09–11 May 1987	20	2 Transects
ALBATROSS IV	ALB87-03	13–20 May 1987	153	2 Surveys
ALBATROSS IV	ALB87-06	18–26 August 1987	71	1 Survey
ALBATROSS IV	ALB88-03	04–06 May 1988	25	3 Transects
ALBATROSS IV	ALB88-10	16–17 November 1988	22	3 Transects
DELAWARE II	DEL89-04	06–17 June 1989	118	2 Surveys
OREGON II	ORE89-04	12–21 August 1989	133	1 Survey

each station site from July 1986 to November 1989 (Fig. 1) (Pikanowski 1995). The observations at the three stations generally were made within one or two days. Water samples were collected one meter above the bottom with a Niskin bottle. Water temperatures were measured with an Omega 866 digital thermistor inserted into the top of the bottle upon recovery. The thermistor was compared each day with an NBS traceable total emersion thermometer housed in a well-stirred cylinder of water. Because the sampling bottle passed through the entire water column both before and after collecting the sample, and was exposed on deck during the actual temperature measurement, the procedure introduced additional possible sampling error. Tests suggest that the accuracy of the measurements is ±0.2°C (Draxler[1]). For each month the three replicate measurements at each of the three stations were averaged to yield a single temperature value for the month.

The oxygen concentration one meter above the bottom was measured each month at station NY6 from 1983 to 1990. From 1985 to 1990, the oxygen concentration approximately 0.1 m above the bottom was also measured at the same time to provide a measure of the near bottom oxygen gradient. During the 12-MDS study (July 1986–August 1989) two observations usually were made on two successive days, and the four values were averaged. During the other periods, generally only a single observation was made. Water samples were collected by Niskin bottles, and the oxygen concentration was determined by a modified Winkler titration method.

Current Meter Temperature Measurements

Current meters deployed in the 12-MDS study were equipped with thermistors to measure water temperature. Even after biofouling interrupted the measurement of currents, the thermistors continued to function properly. Deployments M3 (1987–88) and M4 (1988–89) were within about 5 km of each other (Manning, 1995). When combined, these measurements provide a nearly continuous time series of temperature one meter above the bottom from May 1987 to June 1989. The bottom depth at M3 was 36 m; at M4, 27 m. The accuracy of the temperature measurements is ±0.2°C. The data were averaged to yield a time series of daily bottom water temperatures.

Other Data Sources

The MARMAP sampling program included one station within the inner New York Bight. Station 55 (40°26'N, 73°50'W) is within about 5 km of the replicate stations and the locations of moorings M3 and M4, and it is at approximately the same water depth (30 m). From 1977 to 1987, station 55 was occupied 40 times. Annual cycles of the bottom temperature and the average water column temperature and salinity were calculated from these observations using the method described by Mountain and Holzwarth (1989).

Hourly-averaged observations of surface water temperature, air temperature, and wind velocity were obtained from the Ambrose Tower at the entrance to New York Harbor.

Results

To illustrate and compare the water properties observed in the different surveys, volumetric temperature and salinity distributions were calculated for each survey. While the various surveys covered slightly different areas, volumetric analyses were done for the area north of 40°09'N and west of 73°36'W, which was common to all surveys. The volumetric calculations were made following the method of Mountain and Jessen (1987), using a cell size of 0.5°C and 0.2 Practical Salinity Units (PSU).

Volumetric temperature and salinity distributions for the six complete surveys of the inner New York Bight in 1987 and 1989 are presented in Figures 2a–f. The volume-averaged temperatures and salinities for each distribution are listed in Table 2. The surface and bottom distributions of both temperature and salinity during the two June 1989 surveys are presented in Figures 3–6. Surface and bottom distributions of temperature and salinity, and the vertical distribution of temperature, salinity, and density at each station for all seven cruises in Table 1 are presented in Taylor and Mountain (1991).

Table 2

Volume-averaged temperature and salinity for six surveys of the inner New York Bight. The volumetric distributions of water properties for these surveys are shown in Figure 2a–f.

Dates	Temperature (°C)	Salinity (PSU)
14–17 May 1987	8.27	31.24
18–20 May 1987	8.74	31.43
18–26 August 1987	17.13	31.50
7–10 June 1989	10.72	31.41
15–17 June 1989	12.59	31.26
12–21 August 1989	18.74	30.90

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[1] A. Draxler. NMFS Sandy Hook Laboratory, Sandy Hook, NJ 07732. Pers. comm., June 1991.

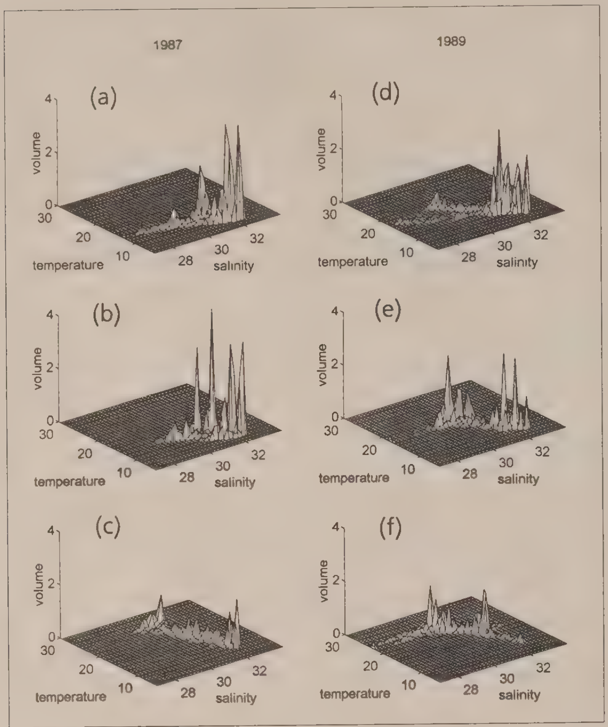
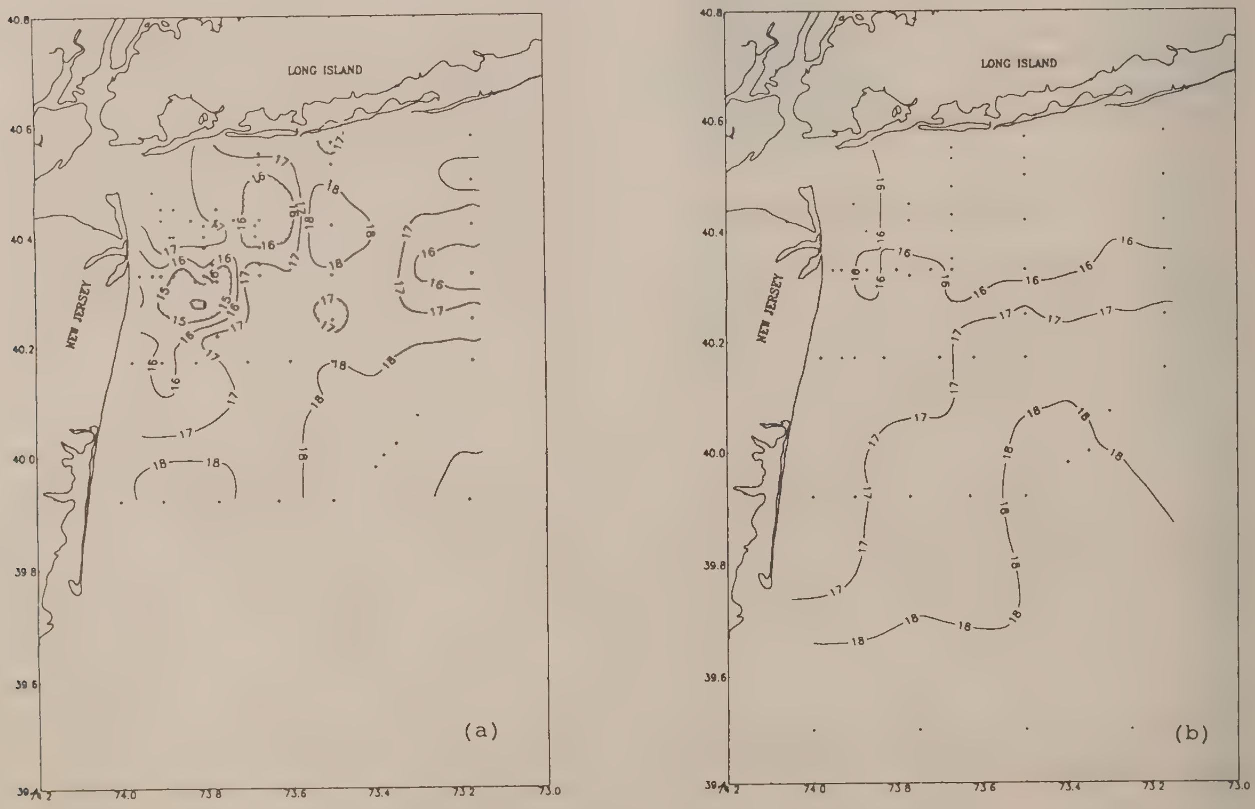


Figure 2
Volumetric temperature (°C)–salinity (PSU) distributions for hydrographic surveys of the inner New York Bight: (a) 14–17 May 1987, (b) 18–20 May 1987, (c) 18–26 August 1987, (d) 7–10 June 1989, (e) 15–17 June 1989, (f) 12–21 August 1989. Volumes in km^3.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(a)
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Figure 3
Surface temperature (°C) distributions for two surveys in June 1989: (a) 7–10 June 1989 and (b) 15–17 June 1989.

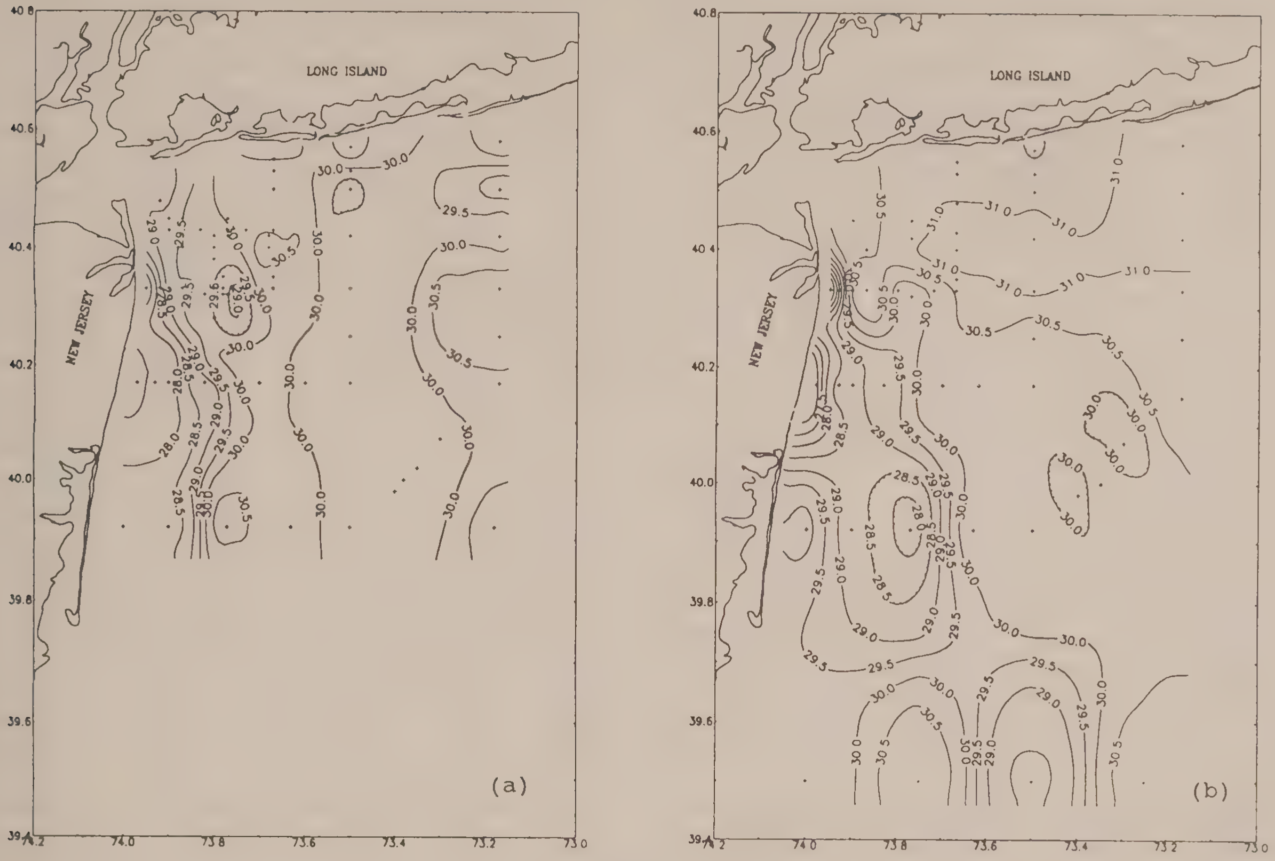


Figure 4
Surface salinity (PSU) distributions for two surveys in June 1989: (a) 7–10 June 1989 and (b) 15–17 June 1989.

The monthly bottom temperatures, averaged from the observations during each monthly replicate sampling at stations R2, NY6, and NY11, are presented in Figure 7 for January 1986 to November 1989. The annual cycle of bottom temperature at MARMAP station 55 is included in Figure 7 as an indication of the long-term, characteristic temperature conditions in the area. The values from the replicate stations generally were similar to the annual curve for station 55 except for the summer of 1988, which was consistently about 2–4°C cooler than the annual curve.

The bottom-temperature time series obtained from the current meter observations in mooring deployments M3 and M4 is presented in Figure 8. The data show that large fluctuations in bottom temperature occured over short periods of time, particularly in summer. The data also indicate that bottom temperatures in the summer of 1988 were 2–4°C cooler than in 1987.

The dissolved oxygen concentration one meter above the bottom at station NY6 generally was high (>8 mg/l) in the winter when the water column was well mixed and low in the summer (<6 mg/l) when the column was stratified (Fig. 9a). The summer minimum value in both 1985 and 1989 was extremely low (<3 mg/l). No trend in dissolved oxygen concentration associated with cessation of dumping in 1987 is evident in the data. The magnitude of the near bottom gradient in oxygen concentration, however, did exhibit a significant trend (Fig. 9b). Before dumping was phased out, oxygen concentration just above the bottom in summer was about 0.5 mg/l lower than the concentration one meter above the bottom. By the summers of 1988 and 1989, the difference in concentration between the two levels was essentially zero.

Discussion

The hydrographic data help provide answers to two questions important in the interpretation of data from

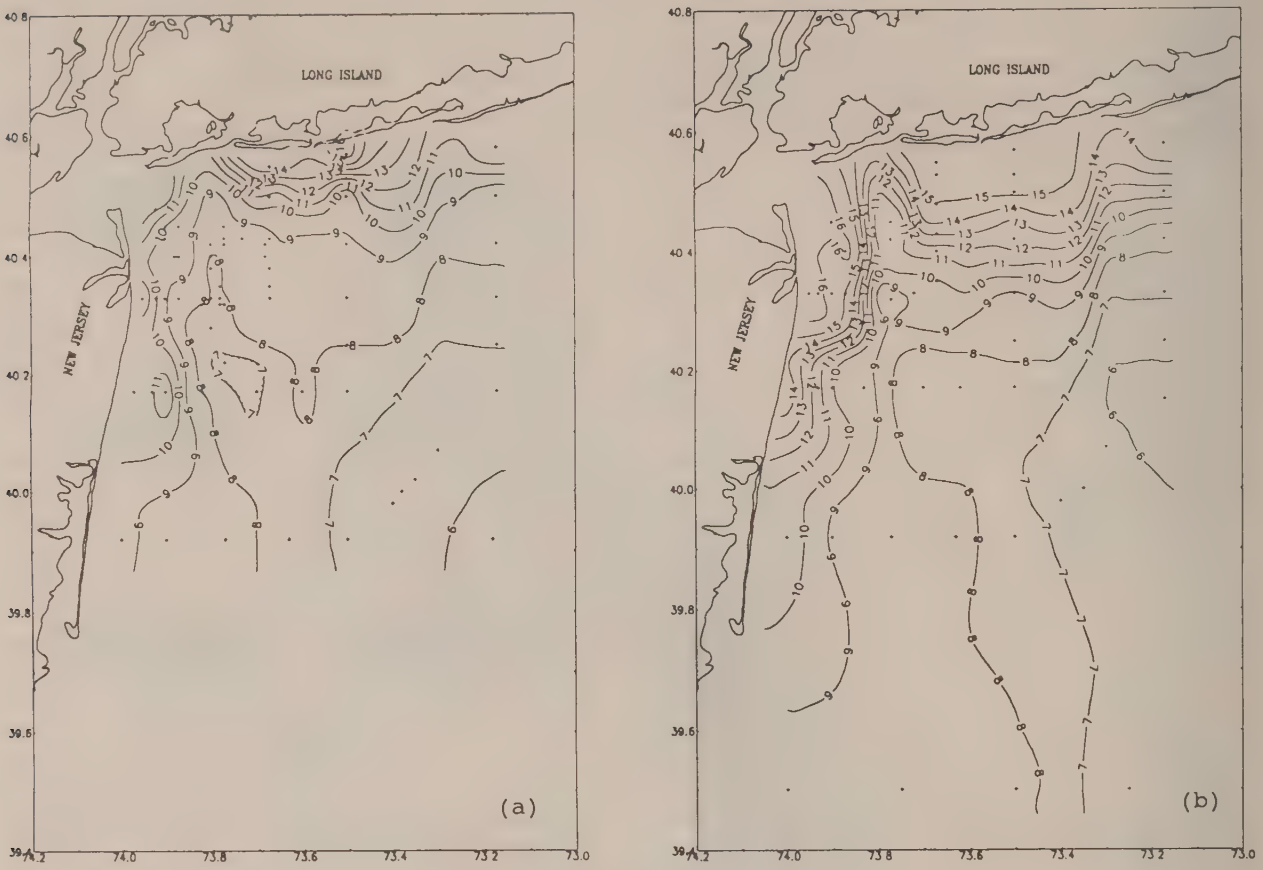


Figure 5
Bottom temperature (°C) distributions for two surveys in June 1989: (a) 7–10 June 1989 and (b) 15–17 June 1989.

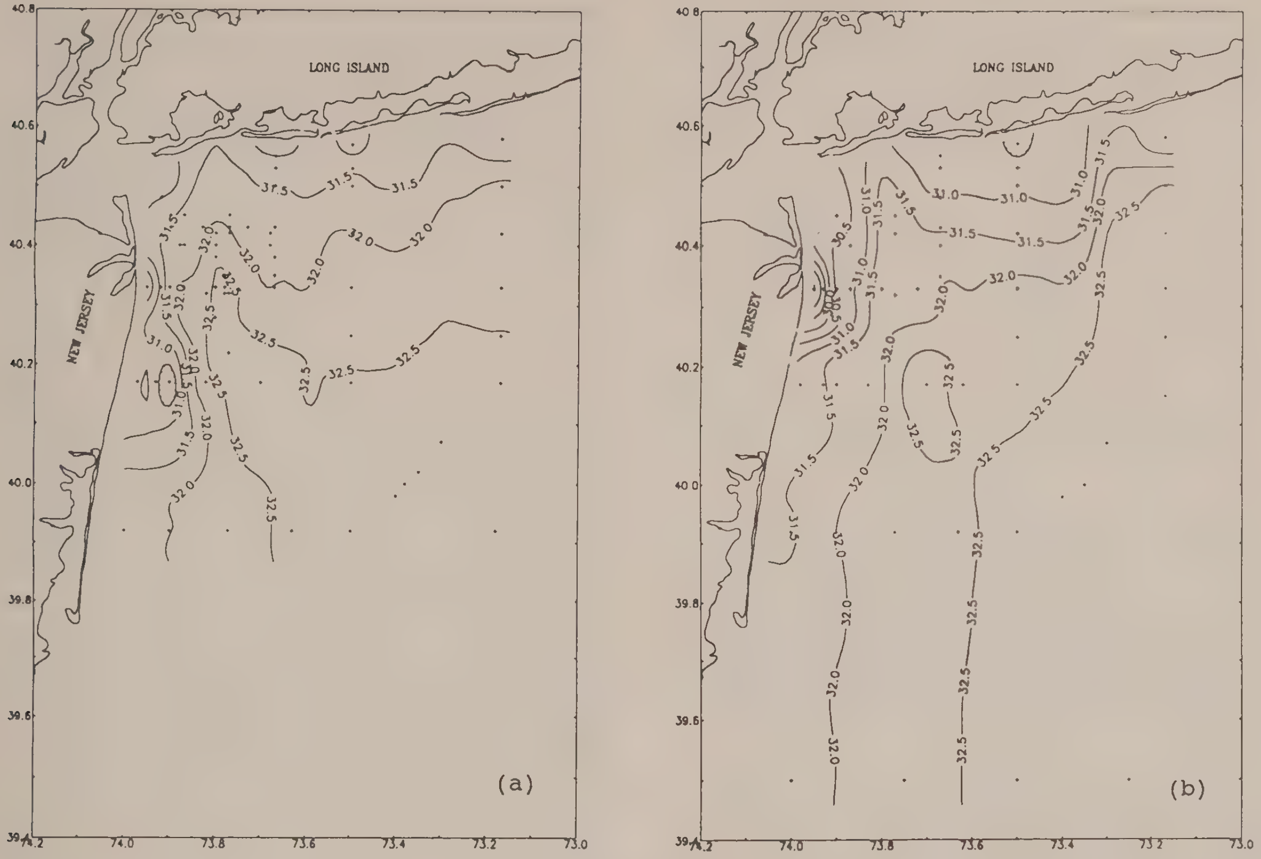
other components of the 12-MDS study. First, what is the temporal and spatial variability of the physical environment in the region around the dumpsite? Second, were the environmental conditions during the dumpsite study representative of the long-term conditions in the region, or were they significantly unusual?

A view of the spatial variability in water properties is provided by the 1987 and 1989 surveys (Table 1). The May–June surveys occurred after seasonal warming had begun, coastal runoff had peaked, and the water column had begun to stratify. The August cruises occurred after stratification was well established. The surface and bottom distributions of temperature and salinity during 7–10 June 1989 (Figs. 3a, 4a, 5a, 6a) illustrate the spatial distribution of water properties under these conditions. While the surface temperature is relatively uniform across the area, the surface salinity distribution distinctly shows the Hudson-Raritan plume as a nearshore band of low-salinity water. As the water moves southward along the New Jersey coast, the plume extends an increasing distance from shore. The salinity contrast between the plume and the other surface waters is greater than 2 PSU. The bottom distributions show increasing salinity and decreasing temperature as the water depth and distance from the coast increase. A tongue of low-temperature, high-salinity bottom water is evident, extending into the northern part of the inner Bight along the axis of the Hudson Shelf Valley.

Data from each of the three sampling programs also provide insight into the temporal variability of water properties in the New York Bight. This variability occurs on short-term (daily or weekly), seasonal, and interannual time scales—each of which is evident in the data sets, as described below.

Short-term Variability

The short-term variability of water properties is best illustrated by the time series of bottom water temperature obtained from the current meters (Fig. 8). Increases and decreases of more than 4°C may occur in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31.5
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Figure 6
Bottom salinity (PSU) distributions for two surveys in June 1989: a) 7–10 June 1989 and b) 15–17 June 1989.

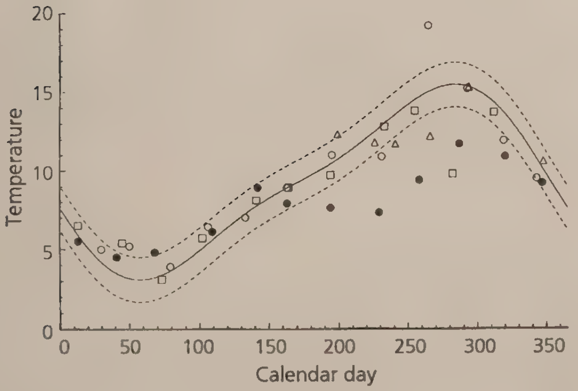

Figure 7
Bottom temperature (˚C) calculated from the observations at the replicate stations R2, NY6, and NY11 for 1986 (△), 1987 (○), 1988 (●), and 1989 (□). The annual cycle of bottom temperature at MARMAP station 55 is indicated by the solid curve, and + and – one standard deviation from the annual cycle are indicated by the dashed curves.

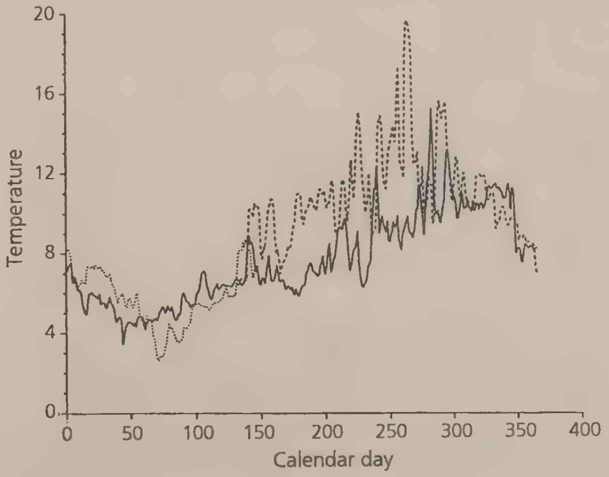

Figure 8
Bottom temperature (˚C) measured by current meters at mooring sites M3 and M4 from May 1987 to June 1989: 1987 (- -), 1988 (—), 1989 (. . .).

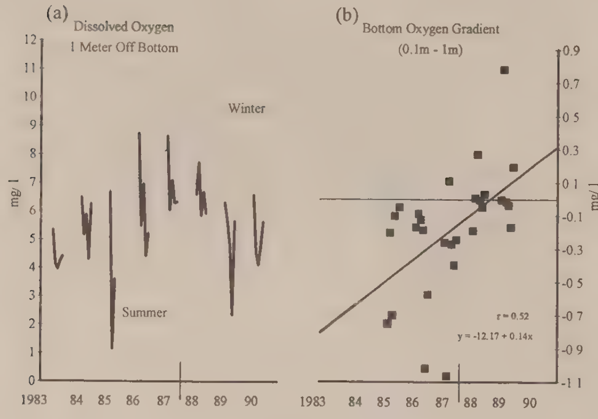

Figure 9
(a) Dissolved oxygen concentration one meter above the bottom at replicate station NY6. The solid line segments represent the stratified season (May–October). (b) The difference in dissolved oxygen concentration between 0.1 m and 1.0 m above the bottom during the stratified season. The dashed line represents the linear regression of the near bottom oxygen gradient against time (years). The regression equation is given in the figure.

less than a day. Most of these events are associated with wind events and the subsequent upwelling or downwelling of water in the region of the Hudson Shelf Valley (Manning, 1991). For example, during the period of largest variability in the 1987 data (calendar days 150 to 300), the change in daily averaged temperature is significantly correlated with the daily averaged eastward wind stress measured at the Ambrose Tower on the previous day ($R= -0.53$, $P<0.01$). Winds toward the east are associated with subsequent decreases in bottom temperature, while winds toward the west are associated with increases in temperature, consistent with wind-driven upwelling and downwelling, respectively (Manning, 1995). In summer, large temperature gradients exist vertically through the water column (the thermocline) and horizontally along the bottom with increasing bottom depth (Figs. 3, 5). As a result, wind-induced displacements of water along the bottom through upwelling or downwelling can result in rapid changes in bottom temperature, as indicated in Figure 8.

Data from the survey cruises illustrate short-term changes in water properties throughout the water column and across the whole inner New York Bight. The two surveys of the inner Bight in May 1987 occurred only a few days apart (14–17 May and 18–20 May). The volumetric temperature and salinity distributions (Figs. 2a,b) indicate little change in water properties over that period. However, the disappearance of the small volume of low-salinity (<29 PSU), high-temperature (>12.5°C) water that was present on the first survey (Fig. 2a) and the appearance of a peak at about 31.4 PSU and 11.5°C on the second survey (Fig. 2b) may have resulted from mixing caused by strong winds before and during the second survey (M. Ingham, NOAA/NMFS, Narragansett, RI, personal communication). The two surveys in June 1989 occurred a week apart (7–10 June and 15–17 June). A comparison of the volumetric temperature and salinity distributions indicates significant change in the water properties between the surveys (Figs. 2d,e). In the first survey a large portion of the volume occurred in a relatively narrow temperature and salinity range and represented the cooler, more saline waters below the thermocline. The warmer and fresher water in and above the thermocline occurred over a wide range of values, but amounted to a relatively small volume of water. In the second survey the volumetric distribution was more bimodal, and most of the low-salinity water (<29.5 PSU) evident in the first survey had disappeared.

Comparing the bottom temperature and salinity distributions of the two June 1989 surveys (Figs. 5, 6) shows an increase in the area of warm, lower salinity water along the coasts and a general southeastward retreat of the isotherms and isohalines between surveys. These changes suggest the occurrence of either a downwelling event between the surveys or an upwelling event before the first survey with relaxation to pre-upwelling conditions between the surveys.

An upwelling or downwelling event could be driven by a strong eastward or westward wind stress, respectively. The wind direction during this period, however, ranged from northeastward to southeastward, with no strong, sustained event to cause significant upwelling or any westward component of windstress to drive downwelling. Also, the current measurements do not indicate the occurrence of either a sustained upwelling or downwelling event (Manning, 1995). No obvious explanation exists for the change in water properties that occurred in the week between the two June 1989 surveys (Figs. 3–6).

Seasonal Variability

The characteristic seasonal change in water properties in the inner New York Bight is indicated by the observations at MARMAP station 55. The annual cycles of water column average temperature and salinity at station 55 are shown in Figure 10. Both the mean cycle and ±1 standard deviation about the mean are plotted. There is about a 14°C range in temperature from the minimum in late winter to the maximum in late summer. The range of the annual cycle in salinity is 0.6 PSU with the maximum in winter and the minimum in summer.

A specific example of seasonal change in water properties is illustrated by the difference in water properties between the first survey in June 1989 and the survey in August 1989 (Figs. 2d, e; Table 2). As expected, the water column was warmer in August than in June. The water column in August was also somewhat lower in salinity. This lowering of salinity probably reflects the continued accumulation in the shelf waters of the spring and summer river runoff.

Interannual Variability

The bottom temperature data from the monthly observations at the replicate stations indicate that 1986, 1987, and 1989 were similar to the long-term average conditions at MARMAP station 55 (Fig. 7). The summer of 1988, however, was 2–4°C cooler than the other years. The time series of bottom temperature measured by the current meters also shows the summer of 1988 to be cooler than 1987 (Fig. 8).

The interannual variability in the inner New York Bight, as a whole, is indicated by comparing the average water property values for the 12-MDS surveys (Table 2) with the water column average temperature and salinity at MARMAP station 55 (Fig. 10). The data in Table 2 represent the entire region and not just one station. Still, the data in Table 2 (plotted as filled circles in Fig. 10) agree quite well with the annual cycle for station 55, falling within one standard deviation of the calculated annual cycles. The average salinity values for the 12-MDS surveys are somewhat low compared to the salinity at station 55 because the surveys include the low salinity Hudson-Raritan plume along the New Jersey coast. No hydrographic surveys of the inner New York Bight, however, were conducted in 1988, when the other data sets considered here indicate unusually cold temperatures.

Unusually cold conditions in 1988 were also reported by Jossi and Benway (1990). Using a long time series of monthly expendable bathythermograph (XBT) transects across the shelf from New York Harbor, they show the summer and fall of 1988 (June to October) to be 2–4°C cooler along the bottom up to 50 km distance from shore. They also show the surface temperatures to be 1–2°C cooler at the same time.

The unusual conditions in 1988 appear to have persisted from about calendar day 150 to day 300 (Figs. 7, 8). The average values of air and water temperature and wind data measured by the current meters and at Ambrose Tower during this period in 1987 and 1988 are listed in Table 3. The bottom water temperature was 2.9°C cooler, the surface temperature was 1.3°C cooler, and the air temperature was 0.5°C cooler in 1988. The smaller differences in the air and surface temperatures suggest that surface atmospheric cooling was not the primary cause of the observed low bottom temperatures in 1988. The average eastward wind stress during the period of interest in 1988 was more than double that of 1987. The greater eastward stress could drive upwelling circulation, which would bring cooler, deeper, offshore water into the Bight. The average bottom current along the axis of the Hudson Shelf Valley was 0.4 cm/s (down valley) in the summer of 1987 and –1.0 cm/s (up valley) in the summer of 1988 (table 1a in Manning, 1995). A wind-driven upwelling would be confined near the coast and not affect conditions on the outer, deeper parts of the shelf, consistent with the observations of Jossi and Benway (1990).

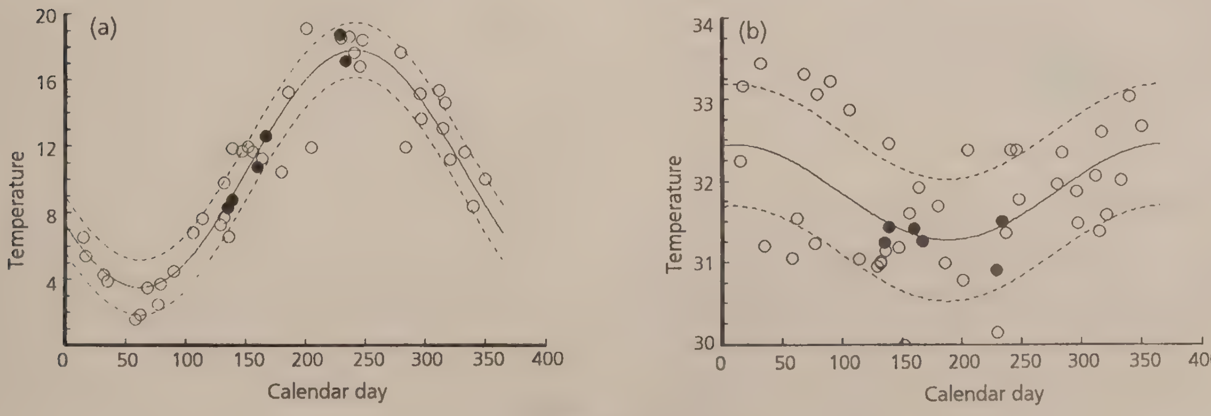

Figure 10

Annual cycle of water column average (a) temperature (°C) and (b) salinity (PSU) observed at MARMAP station 55. The calculated annual cycle for each property is indicated by the solid curve and ±1 SD of the original observations from fitted curve are shown by the dashed curves. The small open circles are the original MARMAP observations. The large, filled circles are the data from the survey cruises in 1987 and 1989 listed in Table 2.

Table 3

Average values for parameters measured at Ambrose Tower and bottom temperature measured by current meters during the period calendar day 150 to day 300 in 1987 and 1988.

	Year	
	1987	1988
Ambrose Tower		
Air temperature (°C)	19.6	19.1
Surface temperature (°C)	19.2	17.9
Eastward wind stress (dynes cm^{-2})	0.24	0.57
Current meter		
Bottom temperature (°C)	11.5	8.6

Near-bottom oxygen values typically decrease in the summer because of the development of density stratification in the water column and the decay by near-bottom processes of settled organic material (Fig. 9a). In the New York Bight, with anthropogenic nutrient loading, this oftens results in hypoxic conditions (Swanson and Parker, 1988). The near-bottom gradient in oxygen concentration is an indication of the rate of bottom decay processes relative to the rate of vertical mixing near the bottom. The summer near-bottom oxygen gradient in the 12-MDS region decreased during the study (Fig. 9b). The existence of the gradient in the earlier years suggests that the seabed oxygen demand was sufficiently large that the time scale of oxygen uptake was shorter than the time scale of mixing within the bottom one meter of the water column. The disappearance of a gradient after dumping had ceased suggests that the seabed oxygen demand was greatly reduced, assumedly because of a reduced flux of material to the bottom, and that mixing maintained nearly uniform conditions in the bottom meter of the water column. This is consistent with the direct measurements of seabed oxygen consumption reported by Phoel et al. (1995).

Conclusions

The oceanographic conditions observed during the 12-MDS study were representative of the characteristic conditions for the area, except for the summer of 1988. From about calendar day 150 to day 300 in 1988 the bottom temperature in the inner New York Bight was 2–4°C cooler than generally observed in other years. The cause of the cooler conditions is believed to be upwelling, driven by unusually strong eastward wind stress. The temperature deviations in 1988 were great enough to have affected some biological and chemical processes studied by other components of the 12-MDS study.

Acknowledgments

We would like to thank the many persons who helped to collect and process the data used in this report. In particular, M. Taylor and T. Holzwarth processed and assisted in the analysis of the CTD data. J. Manning kindly provided the temperature data from the current meters. J. O'Reilly provided the oxygen data and performed the initial analysis showing the decrease in the near bottom oxygen gradient over time.

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Observations of Bottom Currents and Estimates of Resuspended Sediment Transport in the Vicinity of the 12-Mile Dumpsite

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ABSTRACT

Current meter moorings were deployed in the vicinity of the 12-mile sewage sludge dumpsite in water depths ranging from 20 m (near the mouth of New York Harbor) to 53 m (within the Hudson Shelf Valley) and at various times of the year, from July 1986 through June 1989. A total of 10 usable instrument records ranging from one month to one year in duration were obtained. Eight of the ten are near-bottom records.

Both seasonal and geographic variability of wind-induced flow were examined. The wind is most efficient in driving the low-frequency (2–10 day) current during winter when the water column is well mixed and when the eastward component of the wind often induces and sustains an up-valley (northward) bottom flow. Maximum efficiency occurs for wind from 300° (WNW) and at those sites located within the Hudson Shelf Valley.

A continental shelf bottom boundary layer model is applied to estimate resuspended sediment transport. Model inputs include the bottom currents (observed), orbital wave velocities (estimated), and sediment grain size (from the literature). Model output indicates that sediment resuspension occurs approximately 5% of the year, primarily during winter months. Given an along-valley gradient in sediment transport that changed sign depending on the timing and degree of a) wave-induced resuspension and b) wind-induced flow, episodes of both deposition and erosion occurred. The estimate of net deposition/erosion for the Christiaensen Basin (+0.02 mm) for the particular six month focus of investigation was no greater than that for individual storms.

Introduction

For more than half a century, municipalities of New York and New Jersey dumped sewage sludge at a site located approximately 12 nmi south of Ambrose Light Tower, east of Sandy Hook, New Jersey (Fig. 1). In late 1987, after several years of litigation, the site was closed owing to elevated levels of pollutants found in both the sediment and biota (Environmental Processes Division (EPD), 1988).

In July 1986, the National Marine Fisheries Service (NMFS) began a three-year multi-agency observational program to monitor the response of habitat and biota to cessation of sewage sludge dumping (EPD, 1988). Biological, geological, and chemical samples were collected on several space and time scales to assess environmental change.

One component of this multi-disciplinary effort was the deployment of bottom current meters. In an attempt to document storm events that may induce sediment erosion and transport, moorings were anchored at several locations in the vicinity of the dumpsite from July 1986 through June 1989 (Fig. 1). This paper attempts to a) explain the current variability in terms of meteorologic forcing functions and b) make order of magnitude estimates of the resuspended sediment transport.

Much of the pre–1986 physical oceanography pertaining to the New York Bight was conducted during the NOAA Marine EcoSystem Analysis (MESA) Program in the mid to late 1970's. Using data from a set of

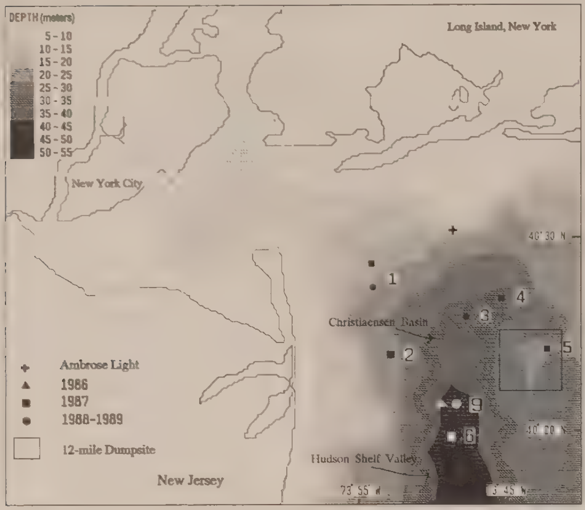


Figure 1

Current meter mooring locations relative to New Jersey, Long Island, the Hudson Shelf Valley, and the 12-mile dumpsite.

current meter moorings deployed for two months during fall 1973 within the upper Hudson Shelf Valley (HSV), Nelson et al. (1978) were the first to present quantitative evidence of wind-induced along-valley flow. They were able to estimate an up-valley flow from October to April and a down-valley flow the remainder of the year by means of a simple model with local wind conditions as input. A more detailed model, including three-dimensionality and both wind- and density-driven dynamics, is presented in Han et al. (1980). Mayer et al. (1982) summarized a more extensive set of current observations, from spring 1974 to spring 1977, in order to present direct evidence of the wind-induced variability: eastward winds driving up-valley flow and westward winds driving down-valley flow. They were able to show differences in the variability due to the cross-shelf location, depth, frequency, and the degree of stratification.

In addition to supplementing the existing MESA database, the data reported in this paper provides time series of bottom currents throughout the year at a location which is closer to the dumpsite near the shallow head of the HSV. While MESA observations of wintertime currents were limited to sites further offshore (LT6 and LT7 in Mayer et al., 1982), our new dataset documents the dynamic and often substantial winter flow that, with the addition of wave-induced resuspension, may govern the annual transport of sediment in the vicinity of the dumpsites. While MESA studies reported bedload transport estimates (Gadd et al., 1978; Lavelle et al., 1978; Vincent et al., 1981), our study focuses on an important mechanism of pollutant dispersion, the resuspended component of sediment transport.

Methods

Data Collection

The vector averaging current meter (VACM) deployments for this study began with a single pilot mooring in summer 1986 (Fig. 1). This mooring (site 9) had three instruments at depths of 14, 28, and 44 m; although the bottom meter failed due to an electronic problem (Table 1). In May 1987, a set of seven moorings with bottom meters (approximately 1 m off bottom) was deployed. Owing to both fish trawling and bio-fouling, parts of the 1987 records were lost, including two complete mooring records. Records were recovered from sites 1, 2, 4, 5, and 6. Three more moorings (sites 1, 3, and 9) were deployed in May 1988, cleaned in November 1988, and recovered successfully in June 1989.

The moorings were deployed near the head of the HSV in water depths ranging from 20 to 53 meters. The sites were chosen based on a) distance to the 12-mile dumpsite and b) direction of local bathymetric gradients. The objective was to document the degree of spatial variability that may be due to topographic and hydrographic conditions. Those moorings closer to the New Jersey shore, for example, were expected to show the effects of the Hudson-Raritan estuarine plume.

Environmental data, collected at offshore NOAA tower and buoy stations, were obtained from either the National Climatic Data Center (NCDC) or the National Oceanographic Data Center (NODC). Hourly average wind velocities were available for the entire study period from Ambrose Light Tower, located a few miles north of the study area (Fig. 1). Surface wave information was also available for the entire study period from either Delaware (Buoy 44012) or Nantucket (Buoy 44008). Surface wave information from Ambrose Light Tower and the Nantucket Buoy, collected concurrently for the first six months of 1990, was obtained as well.

Analyses

Currents—Since low-frequency storm events (2–10 day band) that may transport sediment in or out of the study area are of most interest, the current meter records were filtered to remove the diurnal and semi-diurnal tides (Manning, 1991). A 33-hour low-pass filter described by Flagg (1977) was used. All of the subsequent wind-current analyses, unless otherwise stated, were conducted on the filtered dataset. The raw current meter data (unfiltered), however, are used in sediment transport calculations in order to include the full effect of the tide.

Standard spectral-analysis techniques (Bendat and Piersol, 1986) were applied to the current-meter and wind data to compute the coherence and phase be-

Table 1

Current meter (a) and wind statistics (b) from the 1-hour averaged time series.

Sta #	Water Depth (M)	Instr Depth (M)	North Lat	West Lon	Good Data Start Stop (Time MM/DD/YY)	Eastward Mean (CM/S)	Eastward Std (CM/S)	Eastward Mim (CM/S)	Eastward Max (CM/S)	Northward Mean (CM/S)	Northward Std (CM/S)	Northward Min (CM/S)	Northward Max (CM/S)	Speed Mean (CM/S)	Speed Std (CM/S)	Speed Max (CM/S)
1A	20	19	40 27.6	73 52.2	07/11/87–08/18/87	0.77	4.77	–15	25	–0.32	6.18	–20	36	6.13	4.90	37
B					06/27/88–10/10/88	–0.28	8.70	–25	26	–1.17	11.61	–24	32	13.38	5.73	38
C					11/16/88–06/26/89	–1.04	7.14	–28	28	1.17	10.50	–33	31	11.68	5.22	35
2	22	21	40 23.4	73 52.2	05/09/87–07/08/87	–2.01	6.98	–21	18	–2.24	9.49	–27	21	10.98	5.21	31
3A	36	35	40 25.8	73 48.0	05/06/88–08/31/88	–1.02	5.36	–17	15	0.20	8.03	–18	23	8.89	3.88	24
B					11/16/88–06/26/89	–0.70	5.58	–27	16	5.08	9.81	–23	45	10.56	6.50	45
4	27	26	40 26.4	73 44.4	05/09/87–08/16/87	0.37	6.55	–16	18	–0.42	6.19	–20	17	8.30	3.56	20
5	26	25	40 24.0	73 44.4	05/09/87–07/09/87	–1.01	6.06	–15	20	–2.63	7.98	–26	17	9.46	4.32	27
6	53	52	40 19.2	73 47.4	05/09/87–08/27/87	–0.14	2.32	–10	9	1.08	9.25	–35	25	7.95	5.38	35
9A1	44	14	40 21.0	73 47.4	06/30/86–09/28/86	1.63	10.40	–27	38	3.79	13.75	–40	43	15.40	8.77	46
9A2		28			06/30/86–10/31/86	0.03	5.77	–19	26	5.98	9.81	–20	64	10.08	7.99	65
B		41			06/26/88–06/07/89	–0.72	3.90	–30	13	4.11	13.41	–50	54	12.22	7.94	55

Sta #	Water Depth (M)	Instr Hght. (M)	North Lat	West Lon	Good Data Start Stop (Time MM/DD/YY)	Eastward Mean (M/S)	Eastward Std (M/S)	Eastward Mim (M/S)	Eastward Max (M/S)	Northward Mean (M/S)	Northward Std (M/S)	Northward Min (M/S)	Northward Max (M/S)	Speed Mean (M/S)	Speed Std (M/S)	Speed Max (M/S)
Ambr	25	49	40 30.0	73 48.0	05/01/86–10/30/86	1.56	4.77	–16	13	0.22	4.98	–17	14	6.29	3.23	18
Lght					01/01/87–12/31/87	0.84	5.55	–19	17	–1.10	5.46	–21	20	7.00	3.70	23
Towr					01/01/88–12/31/88	2.45	4.93	–21	17	–0.57	5.82	–20	18	7.15	3.66	22
					01/01/89–06/30/89	0.96	5.62	–22	15	–0.52	5.47	–20	18	6.98	3.73	22
					01/01/90–06/30/90	2.15	5.83	–18	16	–0.16	6.15	–20	18	7.81	3.94	22

tween sites, including both current-current and wind-current relationships.

Winds—Hourly-averaged wind speed and direction, recorded at 49 m above sea level at Ambrose Light Tower, were first filtered to remove spurious data by replacing any values greater than two standard deviations from a 10-hour running mean with that same mean. The filtered values were then converted to wind stress at the air-sea interface according to the method of Large and Pond (1981). Computed wind stress (dynes/cm^2) values were then used to compute wind-current relationships. To test the efficiency of wind in driving the subtidal bottom flow (V), a transfer function (H) was calculated as a linear response to the Ambrose Light Tower wind stress (W):

$$V(\mathrm{f}) = H(\mathrm{f})\, W(\mathrm{f})$$

where f represents frequency. H(f) is non-zero only at frequencies for which the input (W) and the output (V) are significantly coherent.

Waves—Linear wave theory was used to estimate near-bottom wave characteristics. The entire wave spectrum for each hourly observation was supplied by NODC. It was then possible to determine the dominant wave heights and frequencies. Given the water column depth, the resultant root-mean-square (rms) orbital wave velocity and excursion amplitude were calculated for each hourly observation.

Because wave information was not recorded at Ambrose during the deployment period, remote observations were used. A correction factor (gain) and a lead (phase) time were applied to measurements made at the Nantucket Buoy based on six months of observations made during 1990 and were simultaneous with Ambrose measurements. Gain factors of 0.45 and 0.59 for rms orbital velocity and excursion amplitude, respectively, were applied with a 9-hour lead time. These figures were obtained for the 2–10 day band (unsquared coherence=0.64).

Sediments—Sediment samples were taken at several sites on a bimonthly basis during the study period in order to monitor levels of bacteria, toxic metals, and grain size. Because the grain size information was not available at the time of this writing, earlier samples reported by Hathaway (1971) were used as input to the bottom boundary layer model. Reported distributions at each site were proportioned into a maximum of four sediment phi classes (see Table 2).

Sediment Resuspension and Transport—In order to estimate the amount of resuspended sediment transport at individual sites, a bottom boundary layer model (Grant and Madsen, 1979; Glenn and Grant, 1987) was applied to the data from the second and third deployments. Inputs to the model include 1) estimated amplitudes (1.41 times rms) of bottom orbital wave velocity, 2) amplitudes of bottom wave excursion, 3) observed currents at one meter above the bottom, and 4) sediment grain-size distributions. The angle between the currents and the waves is assumed to be zero. The model accounts for the non-linear wave and current interaction, sediment-induced stratification, variable fall velocities for different grain sizes, changes in bottom friction due to the formation of ripples and moveable bedforms, and the associated modifications of velocity in the near-bottom layer. A detailed description of the model is given in Glenn and Grant (1983) and sensitivity studies are reported in Goud (1987). Outputs of the model include estimates of 1) bottom roughness, 2) sediment concentrations above the bottom (categorized according to grain size and distances above the bottom) and, given the estimated velocity profile, 3) the vertically-integrated sediment transport.

Results

Currents

Basic current statistics, i.e. mean, standard deviation, and range at each site, are listed for both the eastward and northward unfiltered raw velocities as well as the vector speed for the entire deployment period (Table 1). The maximum flow is shown to be an order of magnitude greater than the mean and, sometimes, in the opposite direction. At mooring 6, for example, the mean flow is to the north at about 1 cm/s, but the maximum flow was recorded to the south at 36 cm/s. The standard deviations are also much greater than the means.

Time series of the low-pass filtered bottom currents for all deployment periods are plotted along with the estimated eastward wind stress at Ambrose Light Tower (Fig. 2). The two upper water column records (9A1 at 14 m and 9A2 at 28 m) from the 1986 deployment show that as the season progressed into fall, the currents increased. While both instruments recorded primarily northward flow, there appears to be a counter-clockwise turning of current with depth. For the summer of 1987 (1A, 2, 4, 5, and 6), the channelized flow of the deeper HSV site is depicted by the north/south (along-valley) orientation of the lowermost stick plot (6). The large event, for example, on 21 May 1987, indicating strong down-valley flow, is apparent at sites 2, 4, and 5, although in a less intense, multi-directional sense. For the full year deployment of June 1988 through June 1989 (1B, 1C, 3A, 3B, and 9B), the largest events are directed up-valley in the winter and appear to persist for several days. Examples of this persistent up-valley flow can be seen at all three moorings around 1 December 1988 and 8 February 1989.

The distribution of currents are presented with respect to direction (Fig. 3), speed (Fig. 4), and frequency (Fig. 5). Polar bar charts were made for current measured during the full year (June 1988–June 1989) current meter deployment and for wind measured at Ambrose during that same time (Fig. 3). The first column (Fig. 3a) represents the directional distribution (in percent time) of "all" observations while the second (Fig. 3b) and third columns (Fig. 3c) represent the directional distribution of "extremes" during summer and winter. Extremes are defined as percent time of >20 cm/s current and >20 knot (kt) wind. The extreme current at mooring 9, for example, was directed along valley (north or south) for approximately 20% of the

Table 2
Sediment Grain Size Data (Hathaway 1971).

Date mm/dd/yy	Sta #	Lat	Lon	Phi Size % 0	1.5	2.0	2.5	3.0	3.5	4.0
10/25/63	1384	40 14.5	73 43.8	0	0	0	16	22	52	7
07/26/64	1956	40 29.1	73 58.2	24	31	0	0	35	0	11
08/07/64	2007	40 30.1	73 45.9	0	0	38	29	17	16	0

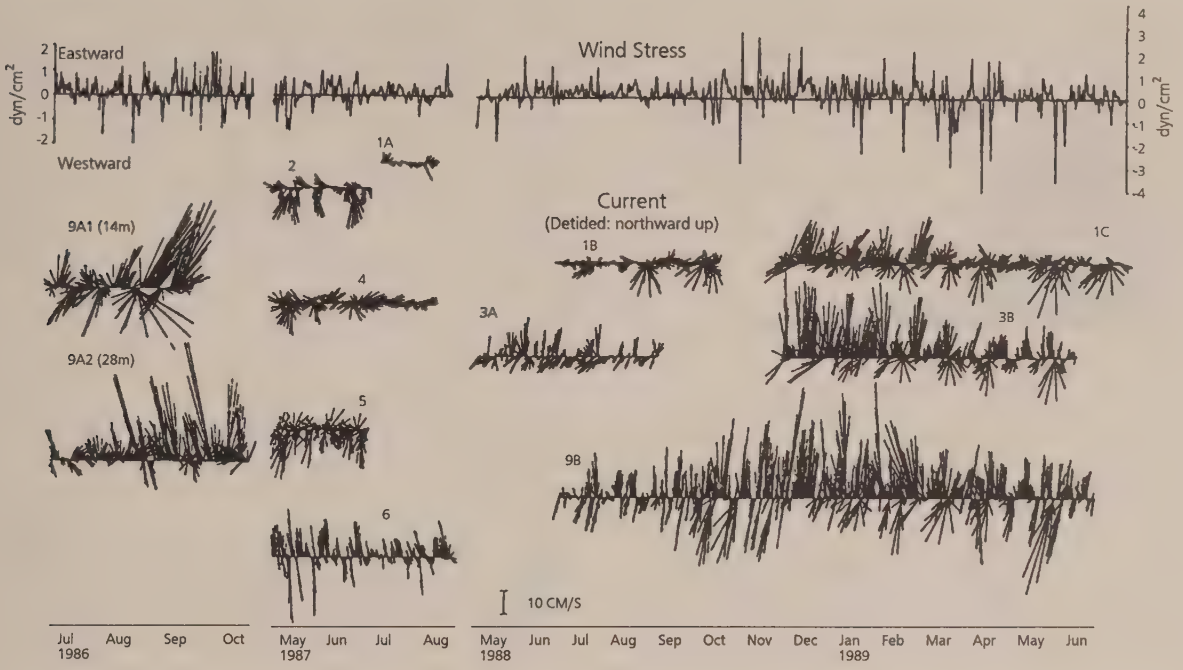


Figure 2
Low-passed current vectors for all current meter deployments (stick plots) and the eastward/westward wind stress from Ambrose Light Tower (top panel) all subsampled four times daily. Eastward wind stress refers to wind blowing from the west off the New Jersey shore.

time. A complete distribution of current speed (unfiltered hourly averaged) vs. percent time for mooring 9 indicates that the peak in this distribution lies in the 5–10 cm/s range due to the presence of tides (Fig. 4). However, it is the less frequent extreme currents that are the focus of this investigation. The distribution of current variability vs. frequency is presented in terms of spectral density (Fig. 5). While the spectrum peaks as expected for the semi-diurnal tides and its harmonic, the highest energy is associated with the lower frequency (wind-driven) cycles.

Progressive vector diagrams for all three current meter deployments are plotted (Fig. 6). (To fit the complete record, the scale is different from the mapped area.) The two good records from 1986 indicate a northeastward and northward flow for the surface (14 m) and mid-depth (28 m) layers, respectively, with most of the distance covered during the last two months, September and October (Fig. 6a). Records from summer 1987 demonstrate the spatial variability that may be observed over one to three summer months and the relatively stagnant conditions of that season (Fig. 6b). In the final deployment shown in the rightmost panel, mooring records 1C, 3B, and 9B depict strong northward flow from December through February (Fig. 6c). At site 9B, for example, approximately 500 km of water

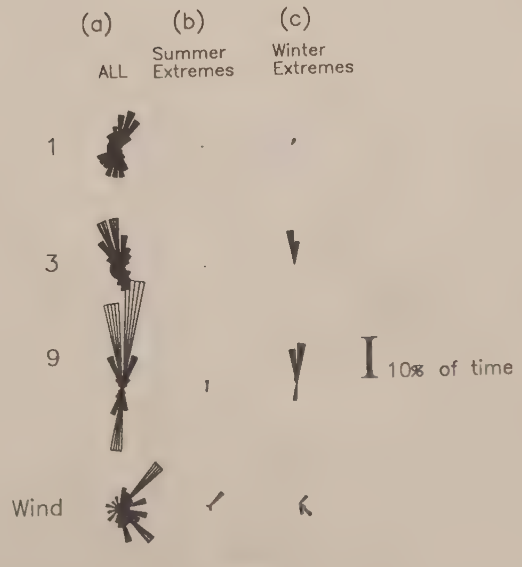


Figure 3
Distribution of unfiltered current with respect to direction at sites 1b, 1c, 3a, 3b, 9b, and Ambrose Light Tower for June 1988 through June 1989 including (a) all, (b) extreme summer, and (c) extreme winter observations. "Extreme" represents observations of >20 cm/s current and >20 kt wind. The arrow, as shown in legend on the right, refers to % time. Extreme current is usually northward and occurs in winter.

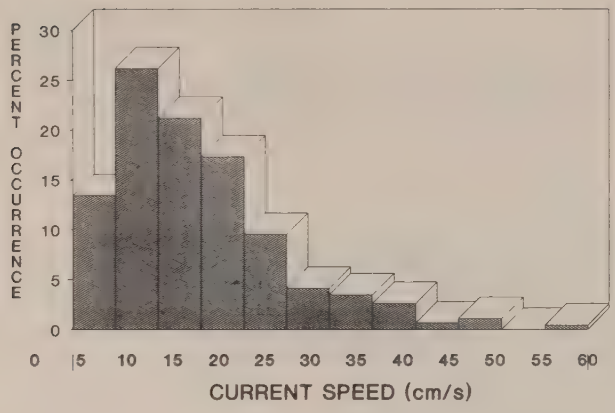


Figure 4

Distribution of unfiltered current speed at site 9 for the full year June 1988 through June 1989.

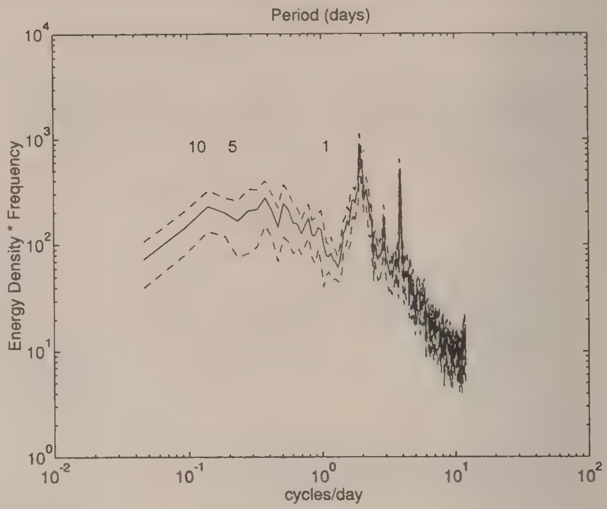


Figure 5

Distribution of unfiltered current frequency at site 9 for the full year June 1988 through June 1989. Power spectral density units are $(cm/s)^2/cpd$. cpd=cycles per day.

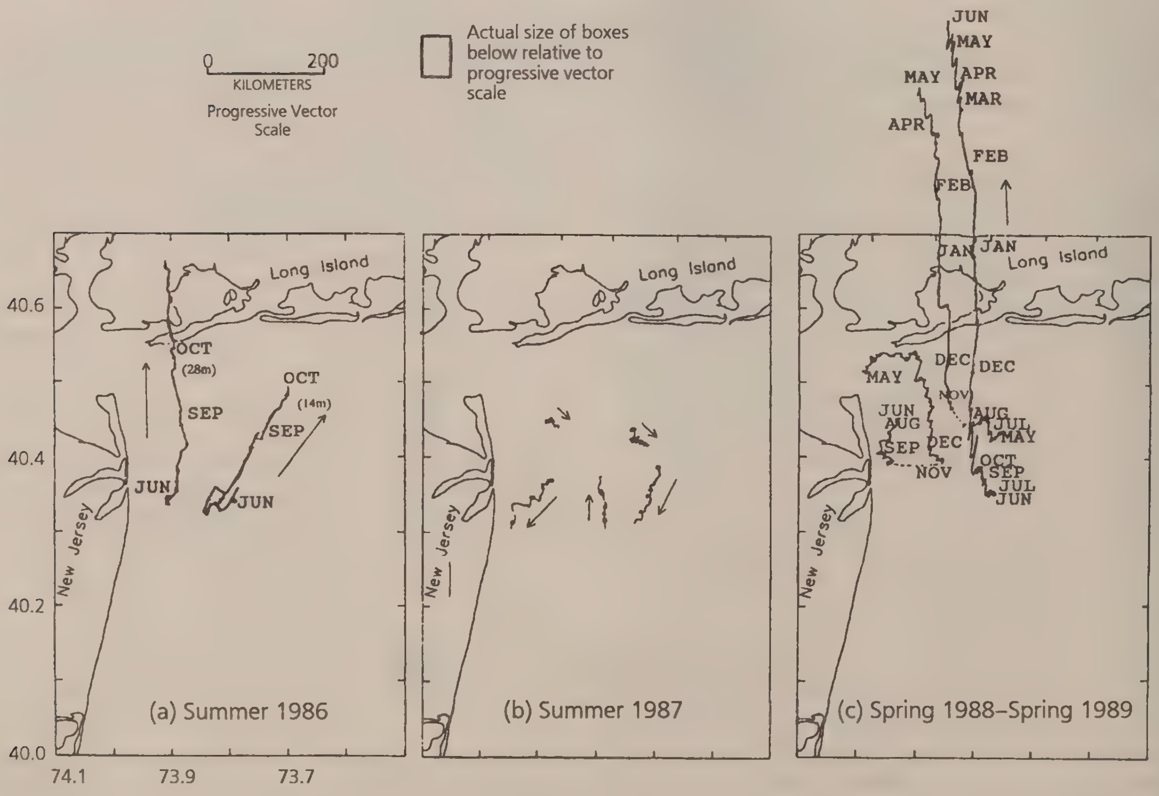

0
200
KILOMETERS
Progressive Vector Scale
Actual size of boxes below relative to progressive vector scale
Long Island
New Jersey
OCT
(28m)
SEP
JUN
OCT
(14m)
SEP
JUN
JUN
MAY
MAY
APR
APR
MAR
FEB
FEB
JAN
JAN
DEC
DEC
MAY
NOV
JUN
AUG
AUG
JUL
MAY
SEP
DEC
NOV
OCT
SEP
JUL
JUN
40.6
40.4
40.2
40.0
74.1
73.9
73.7
(a) Summer 1986
(b) Summer 1987
(c) Spring 1988–Spring 1989

Figure 6

Progressive vector diagrams for (a) summer–fall 1986, (b) summer 1987, and (c) spring 1988–spring 1989. Arrows indicate the general direction of flow. Note that vectors have a different scale than the map area, and the starting point for 1986 mooring is offset to prevent clutter (see Table 1 for dates data were collected).

flowed by the spot in three months. Apparently, there is a region of strong divergence north of the study area.

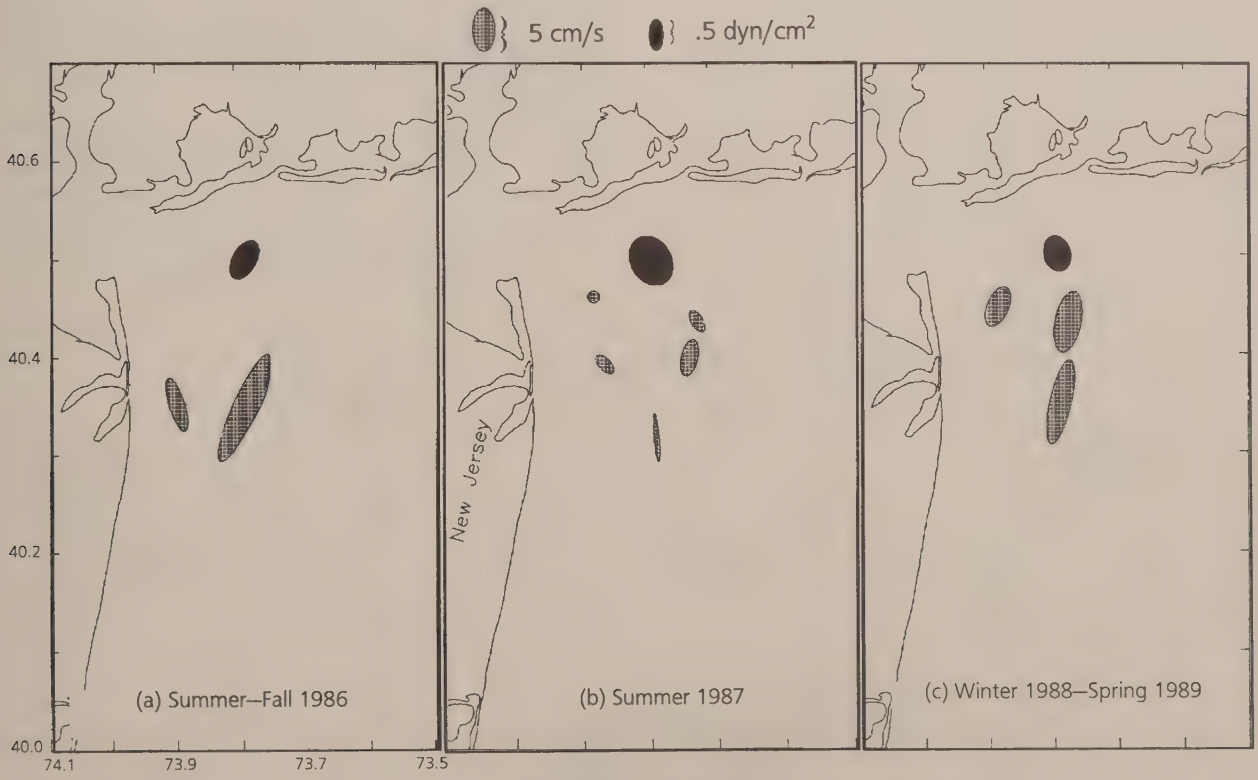
The current ellipses are depicted with the length of their axes associated with the standard deviation of flow in the respective directions (Fig. 7). Filtered current data was used in this calculation. The wind stress variability, represented by the hatched ellipses, is shown to be multidirectional (low ellipticity) unlike the current variability which is shown to have a larger N/S component.

The coherence and phase of current meter deployments 1C, 3B, and 9B is presented for the 2–10 day band (Table 3). The values above the diagonal refer to the (N/S) flow and those below the diagonal refer to the cross-valley (E/W) flow. While the along-valley flow is fairly coherent among the three mooring locations, the cross-valley flow coherence is far less. Consequently, while the cross-valley flow is small compared with the along-valley flow, it is evidently more sensitive to the different bathymetric gradients. The phases computed for each pair (relative to the 2–10 day band) are given in hours where, for example, the value "9.5 (5.8)" indicates that the east/west flow at mooring 9B leads that at mooring 1C by 9.5 hours ± 5.8 hours with 95% confidence. Since this is the largest lag(-)/lead(+) between moorings, it appears that the mooring sites respond almost concurrently within this frequency band. Table 4 includes the coherence, gain, and phase between the eastward wind stress and the northward flow at all sites. Values are listed for low-frequency (<0.5 cycles/day) currents only because higher frequency relationships were not significant.

Sediment Transport

Estimates of resuspended sediment transport are given in Table 5. The units are vertically integrated sediment concentration times velocity. This may be written as (cm^3 of sediment/cm^3 of water) × (cm/s) layer thickness which reduces to cm^2/s.

The results indicate a very sporadic occurrence of transport that occurs primarily during the winter months and most often towards the north. As discussed below however, the "mean" values are the result of only a few



(a) Summer—Fall 1986

(b) Summer 1987

(c) Winter 1988—Spring 1989

Figure 7

Directionality of current variability represented by the current ellipses for (a) summer-fall 1986, (b) summer 1987, and (c) winter 1988-spring 1989. The length of the major and minor axes of each ellipse correspond to the standard deviation of the flow in their respective directions. The hatched ellipses are associated with the wind stress for the Ambrose Tower site.

events that are orders of magnitude larger than other events and may not be representative of the long-term character of the entire region.

Nevertheless, both the episodic (event-driven) and seasonal nature of the estimated transport can be depicted in a stick plot presentation (Fig. 8). The uppermost panel is the orbital wave velocity measured off Delaware and Nantucket for the second and third deployments, respectively (Fig. 8a). (Note that values <20 cm/s are not plotted.) In order to depict more than a few large events, it was necessary to log-transform the sediment transport data (Fig. 8b). (Note that values less than 10^{-7} cm^2/s are not plotted.) During the second deployment there was evidently a single event in late May 1987, mentioned above, during which the orbital wave velocities exceeded 20 cm/s simultaneous with a strong downwelling event (Fig. 2). The third deployment documents another relatively quiet summer period followed by several up-valley (northward) transport events that occurred in the winter of 1988–89. If the arithmetic values for sediment transport are plotted in this way (rather than order of magnitude), the smaller, multi-directional events, such as those observed in October 1988, appear insignificant compared with the up-valley winter events. Whether they are indeed “insignificant” or not is left for the discussion below.

Table 3

Coherence and Phase (hours) between current meter mooring sites for 2–10 day band. Values above and below the diagonal refer to along-valley (N/S) and cross-valley flow (E/W), respectively. Values within parenthesis refer to the 95% confidence interval.

Station		1C	3B	9B
1C	Coh.		.83 (.33)	.83 (.33)
	Phase		–5.5 (2.8)	–1.8 (2.5)
3B	Coh.	.54 (.33)		.94 (.33)
	Phase	–1.8 (6.3)		3.0 (1.3)
9B	Coh.	.57 (.33)	.63 (.33)	
	Phase	9.5 (5.8)	8.5 (5.0)	

Discussion

Wind-Current Relationship

Coherence estimates indicate that nearly half of the low-frequency current variability is explained by the wind. Gain factors range from approximately 4.0 at mooring 1 to 13.0 at mooring 6, indicating that the wind forcing is several times more efficient at the deeper, HSV site. Phase factors range from 0 to 85 degrees, indicating that there is a quick setup time (usually less than one day) between the wind and current velocity in the 2–10 day frequency band.

To determine the wind direction that was most efficient in driving the along-valley flow, the coherence and phase between the flow and wind rotated around the compass in 15° intervals were calculated. The results demonstrate the effectiveness of winds from the west-northwest (or towards the east-southeast) (Fig. 9).

Table 4

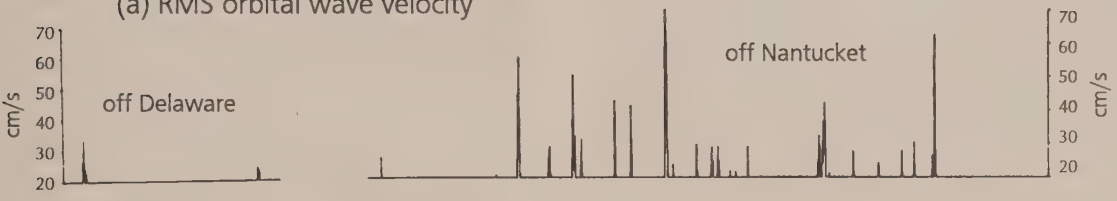
W/E wind stress and N/S current relationship averaged over 2–10 day cycles.

	Coherence		Transfer Function				
Mooring	Estimate	95% Confidence Level	Gain	Confidence Interval	Phase (deg.)	Confidence Interval	Record Length (days)
1A	.77	.44	4.3	1.7	29.	27.	27
1B	.51	.38	3.1	2.2	65.	25.	103
1C	.71	.17	5.0	.9	50.	20.	220
2	.78	.34	7.4	2.2	37.	14.	49
3A	.48	.24	6.5	2.9	78.	28.	114
3B	.59	.24	6.5	1.7	85.	26.	187
4	.73	.26	5.2	1.3	9.	16.	88
5	.70	.34	9.2	3.4	0.	14.	55
6	.66	.24	13.0	3.8	3.	15.	88
9B	.63	.13	11.6	1.9	71.	23.	337

Table 5
Estimates of Vertically–Integrated Sediment Transport in (cm^2/s).

		Eastward				Northward			
Sta	# pts	Mean	Std	Min	Max	Mean	Std	Min	Max
1A	837	0.000000	0.000000	–0.000000	0.000000	0.000000	0.000000	–0.000000	0.000000
1B	1660	–0.000003	0.000045	–0.001633	0.000067	0.000002	0.000034	–0.000003	0.001165
1C	5283	–0.001448	0.093110	–6.759000	0.008456	–0.000142	0.008543	–0.602000	0.023240[1]
3A	711	0.000000	0.000000	–0.000000	0.000001	0.000000	0.000000	–0.000000	0.000006
3B	4501	–0.000194	0.002481	–0.054550	0.001362	0.000187	0.003582	–0.038760	0.063280[1]
4	2293	0.000000	0.000005	–0.000108	0.000060	–0.000007	0.000111	–0.003973	0.000000
5	1384	–0.000004	0.000039	–0.001033	0.000000	–0.000010	0.000103	–0.002605	0.000001
6	2568	0.000019	0.000229	–0.000088	0.005868	–0.000325	0.003781	–0.089830	0.000004
9B	7427	–0.000011	0.002537	–0.033610	0.072400	0.001506	0.022900	–0.088630	0.565000[1]

[1] Includes winter observations.


(a) RMS orbital wave velocity

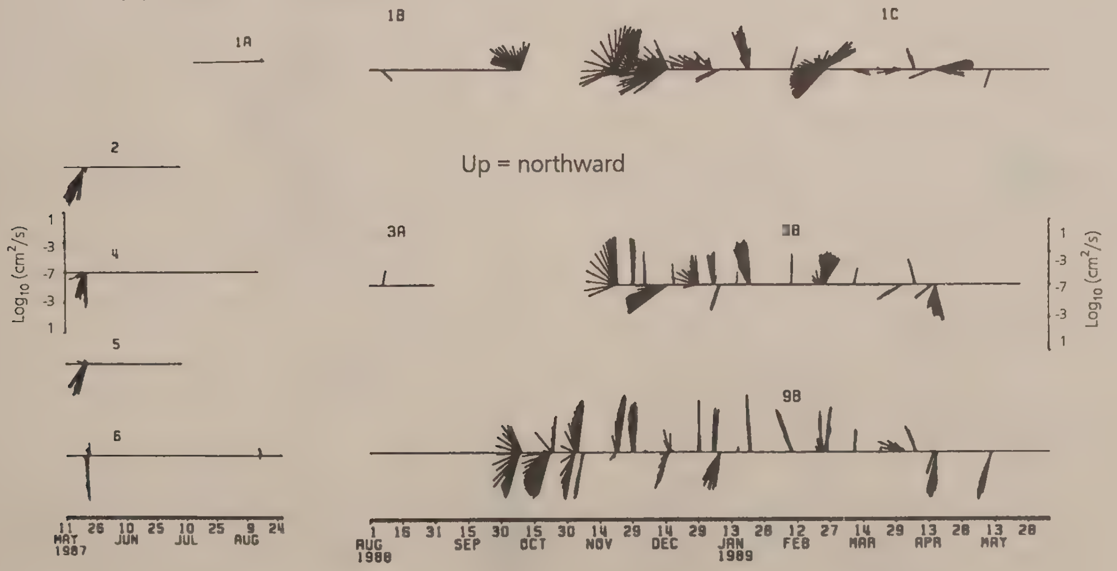

(b) Sediment transport

Figure 8

(a) Root-mean-square (RMS) orbital wave velocities (>20 cm/s) and (b) sediment transport estimates (log$_{10}$ [cm^2/s]).

To determine whether the winds recorded during the study period were representative of the historical mean wind, the winds recorded at Ambrose Light Tower (1987–89) were compared with those recorded at New York City (NYC) and JFK Airport (JFK) (1948–65; NCDC[1]). Using a definition of "strong" winds as those exceeding 11 kt at NYC or JFK and 12.2 kt at Ambrose, strong winds occurred approximately 11% more frequently (52% vs. 41% of the time) during the study (Fig. 10). It is impossible to claim a significant difference in the magnitude, however, since the relationship between the three different recording stations (NYC, JFK, and Ambrose) is uncertain. Given the difference in the long-term mean of the two datasets, a gain of 1.11 (Ambrose, NYC, and JFK) was used nevertheless. While winds during the study period included anomalous strong southwesterlies, most of the time, as in the historical case, strong winds came from the northwest quadrant, which are most effective in driving upvalley flow.

Resuspended Sediment Transport

Determining the dominant forces governing bottom flow is only the first step in a multi-phased project to estimate sediment transport in and around the 12-mile dumpsite. The estimated orbital wave velocities, an important sediment resuspension mechanism, along with low-frequency bottom current records at all sites (except for summer 1986 when near-bottom current was unavailable) were input to an empirical bottom boundary layer model.

Before interpreting the results (Table 5, Fig. 8), there are several considerations to take into account. The first is that since the standard deviations of transport estimates are at least an order of magnitude greater than the means, it is impossible to claim that the mean values are representative of the long-term (climatologic) mean. The "means" calculated in this study are highly influenced by a few events that may or may not be characteristic of the area. The single storm event which exhibited 71 cm/s orbital wave velocities on 14 December 1988, for example, resulted in maximum westward transport of 6.8 cm^2/s at mooring 1C. This single event produced a "mean" transport over the entire deployment period that was larger than the mean at any other site by an order of magnitude. Only after a careful use of the historical wind data and both wind-current and wind-wave relationships (beyond the scope of this report) will it be possible to build a dynamical model accurate enough to integrate and define the long-term sediment transport.

A second consideration is that these vertically-integrated transport estimates are not the same as the net sediment transport. Estimates reported in this paper simply represent a "flux" of sediment that passed a single point. The spatial gradients of these transport estimates, however, give us the deposition/erosion rate.

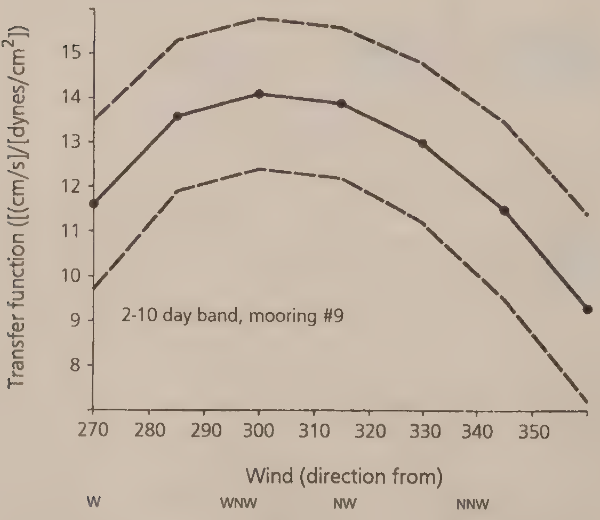


Figure 9
Transfer function ([cm/s]/[dynes/cm^2]) of the wind-current relation for mooring 9, with one point per 15° of the compass. Dashed lines represent the 95% confidence limits.

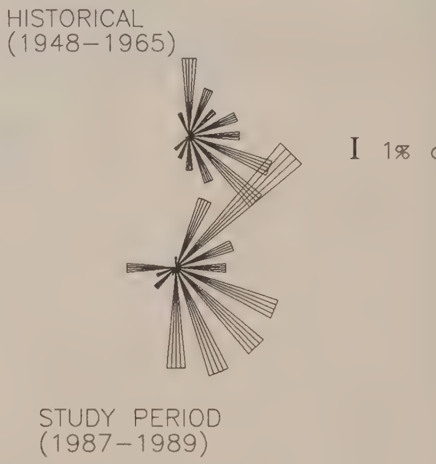


Figure 10
Directional distribution of historical (1948–1965, New York City [NYC] and JFK airport [JFK]) strong winds vs. study period (1987–1989, Ambrose Light Tower) strong winds. "Strong" winds for NYC and JFK are those greater than 11 kt. "Strong" winds for Ambrose Light Tower are those greater than 12.2 kt. A gain factor of 1.11 came from the difference in the long-term mean wind speeds of the two data sets. Most strong winds are from the northwest towards the southeast.

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[1] NOAA, NESDIS, National Climatic Data Center, Asheville, NC 28801–2696, July 1991.

In order to obtain a more realistic representation of the spatial gradients (i.e. divergence), it will be necessary to apply numerical box modeling techniques to a dense station grid that accurately represents changes in oceanographic and bathymetric environments. Spatial derivatives (due to the changes of surficial sediments, orbital wave velocities, and currents) are needed to accurately estimate sediment deposition/erosion throughout the region. A preliminary investigation of deposition/erosion is presented in a subsequent section.

Finally, the sediment transport estimates have not included a correction for a depth of erosion limit. As the fine grain deposits are depleted during a particular event, there is inevitably a change in the magnitude of resuspension as the composition of the bedload is modified to include mostly larger sand grains. (Estimates of bedload transport are not included in the numbers reported here.) It is therefore possible that the smaller events depicted in the stick plot (Fig. 8b) may be as "significant" as the larger events in terms of reworking the available fine-grained material.

These results do suggest, however, the frequency, the direction, and the order of magnitude transport to be expected at the mooring locations themselves. Although several sensitivity tests were conducted (i.e. alterations were made in the model input variables such as a) not applying the nine-hour phase lag for Ambrose vs. Nantucket waves, b) allowing sediment grain size distributions to be 100% silt, c) applying a correction factor for the cohesiveness of the sediment, and d) using non-filtered current velocities), the final results were affected by less than an order of magnitude. Since the spatiotemporal variability of the estimates spans several orders of magnitude, it appears that the model is sufficiently robust for purposes of this study.

The seasonality of sediment transport (Table 5) is in fact what one might expect from the wind-current relations described above. Estimates from moorings deployed during the winter of 1988–89 (1C, 3B, and 9B) are at least an order of magnitude larger than the estimates from summer deployments. (The only exception to this observation is the case of mooring 6 which, because of a single down-welling event in late May 1987, has a mean southerly transport of 0.000325 cm^2/s.) The model estimates zero transport, in fact, for the month-long deployment at 1A in summer 1987.

The conclusion that winter storm events dominate the overall annual transport is in agreement with other recent findings on the northeast continental shelf. Tracer studies conducted in this area (Lavelle et al., 1978) found that for over 135 days in the fall of 1973 nearly 90% of the transport occurred within a single two-day December storm.

Event Analysis

It is instructive to take a closer look at a few of the "events" that are so important to the long-term sediment transport, beginning with a wind event (Fig. 11a). When the eastward winds apply stress that moves the surface waters to the right (southward), a compensating return flow at depth (dashed line in Fig. 11b) is directed up the HSV to the north with a lag time of approximately 18 hours. It so happened that, for the up-valley events on 29 December and 4 January and the down-valley event of 7 January, the surface waves simultaneously generated orbital velocities at the bottom on the order of 20 cm/s (solid line in Fig. 11b). Synergism resulted in considerable sediment transport (Fig. 11c). If, however, the surface wave energy had been completely out of phase with the along-valley flow event, little or no transport would have occurred. The events shown have a typical duration of less than 24 hours.

Deposition/Erosion

As mentioned earlier, there is an important distinction to be made between the sediment transport estimates and actual deposition or erosion at individual sites reported in this paper at the respective sites. A simple box model was used to calculate the important cross-shelf gradients. Increasing water depth in the offshore end of the box for example (Fig. 12), decreases wave-induced resuspension which, in most cases, tends to decrease the amount of sediment transport. (Because we are only interested in cross-shelf gradient in this highly idealized model, the alongshelf gradients are set to zero so that the model "boxes" are actually two-dimensional rectangles.) In this way it is possible to make estimates of deposition or erosion since what comes into one side of the box is not the same as what exits from the other side. The difference is what gets deposited or eroded to or from the bottom. Since the errors that have accrued thus far in the transport modeling processes likely exceed any estimates of net sediment transport, this last calculation is not attempted here. A single example is presented, however, to make estimates of deposition/erosion.

A single box extends from mooring 3 (36 m) to mooring 9 (44 m) over a 8.5-km cross-shelf distance. Given the characteristics of this idealized box representing the area of the Christiaensen Basin, the model estimates episodes of both deposition and erosion. While deposition slightly exceeds erosion during this period of investigation (November1988–May 1989) by 0.02 mm, the difference is no greater than the storm-to-storm variability.

Three different storms were examined in detail. On

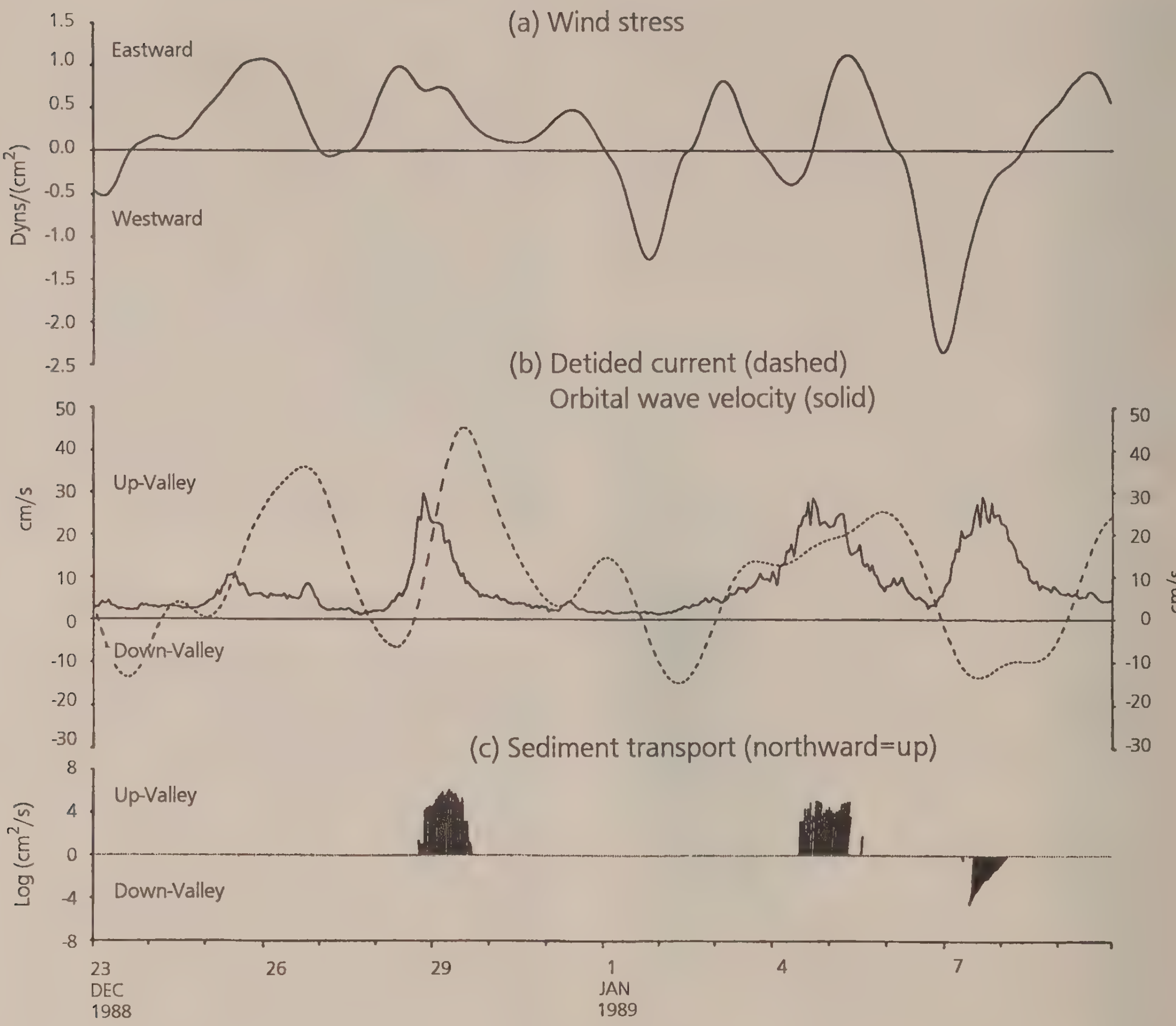

Figure 11

Example of three sediment transport events including (a) the wind stress, (b) the root mean square (RMS) orbital wave velocity and the subtidal flow near the bottom, and (c) estimate of log-transformed resuspended transport in cm^2/s.

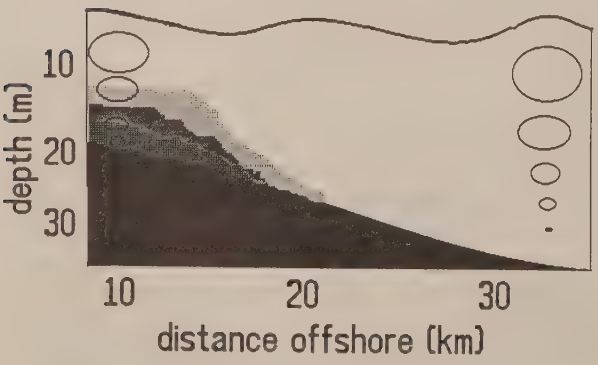

Figure 12

Schematic of cross-shelf differences in wave-induced resuspension with the lighter stippled area representing resuspended sediment especially heavy in the shallower sites. Cross-shelf gradients in wave-induced resuspension and other model input parameters are needed to estimate net sediment transport (see text).

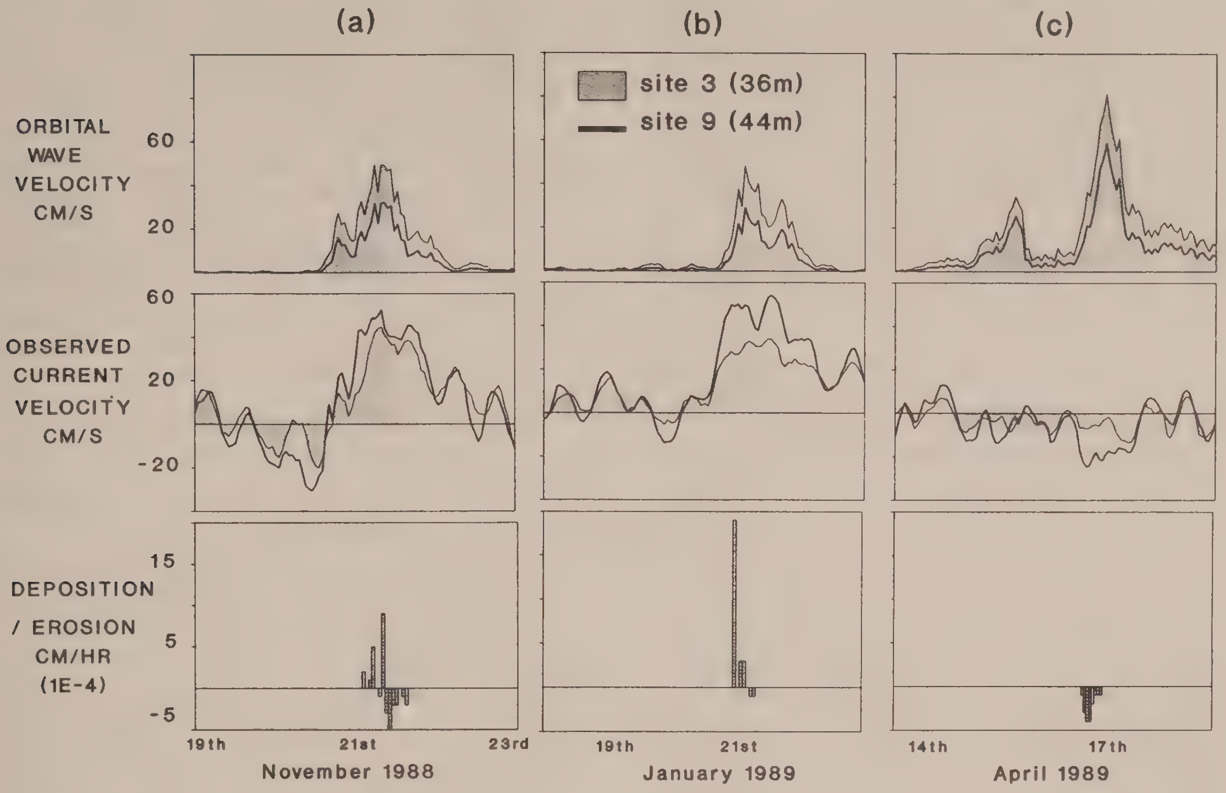

Figure 13
Examples of (a) mixed, (b) depositional, and (c) erosional events due to cross-shelf gradients in orbital wave velocity and observed currents.

21 November 1988, the model estimates an event (Fig. 13a) that was at first depositional and later erosional (total = +0.01mm). During the first half of this storm, the observed current was significantly less at mooring 3. Despite the larger orbital wave velocities at mooring 3, the sediment was deposited because the sediment flux leaving the basin was less than that entering. Later in the storm, however, the sediment was eroded because, while the current speeds at the two locations were similar in magnitude, the increased wave-induced resuspension at the shallower site tended to carry more sediment out of the basin than that which entered. The storm that occurred on 21 January 1989 (Fig. 13b) was a depositional event (total = +0.021 mm) while that which occurred on 14 April 1989 (Fig. 13c) was an erosional event (total = –0.0016 mm).

If the orbital wave velocity decrease offshore was the only cross-shelf gradient of input parameters, one would expect deposition and erosion during periods of down-valley and up-valley flow, respectively, but, as close examination of these events indicates, the cross-shelf gradient in bottom flow may be important as well. In order to resolve the spatial variability of the model input parameters, it will be necessary to implement the three-dimensional circulation models (wind-driven, time-dependent) currently under development at various agencies (Blumberg and Galperin, 1990; Scheffner et al., 1991; Oey[2]).

Conclusions

The subtidal flow field in the vicinity of the 12-mile dumpsite is coherent among sites, especially for the along-valley flow (0.83–0.94) and less so for the cross-valley flow (0.54–0.63). Phases between sites are all less than a half day for the 2–10 day storm band.

West-northwest winds, dominant during the winter months, are most efficient in forcing northward flow both within the Hudson Shelf Valley (40–60 m) and at sites on the shallower flanks (20–40 m), often in excess of 25 cm/s for several days duration. During the remainder of the year, the magnitude of the flow is approximately half as large and the direction of flow is more variable.

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[2] Oey, L-Y. 1991. Development of a three-dimensional model for prediction of transport of accidental spills and hazardous waste through New Jersey coastal waters. Year II: final report to the New Jersey Dept. of Environmental Protection.

A continental shelf bottom boundary-layer model was applied to the observations of bottom currents, bottom wave characteristics, and sediment grain size data in order to obtain order of magnitude estimates of resuspended sediment transport at all sites. The most significant conclusions in terms of sediment transport are that a) very few events contribute to the annual transport at any one site, b) the resuspended transport estimates are dependent on the simultaneous occurrence of wave-induced resuspension and wind-induced flow, c) the frequency and magnitude of the transport increases in the winter and is usually directed up-valley, and d) the transport of sediment past a single location is not the same as the net sediment transport.

Only with careful attention to the direction of bottom flow events, the statistical probability of flow events occurring simultaneously with wave-induced resuspension events, and the spatial gradients of these events throughout the inner New York Bight, may one continue the very difficult task of estimating the net transport of sediment in this very complex environment. It will be necessary to apply a three-dimensional circulation model that may eventually be coupled to both surface wind-wave and bottom boundary-layer models.

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Audience Questions

Question: The theme of this conference is recovery, and part of what controls recovery is the turnover time of the water column. Based on your model, can you give us an idea of what the turnover time is of water north of NY11 (the reference site) or somewhere around your station 3?

J. Manning: Approximately one or two times per month, a strong wind-induced event could cause an advection of the water mass outside of the study area, particularly in the wintertime. Depending also on the position in the water column, a parcel in the upper layer can be advected more easily. In the bottom layer, depending on which way the wind is blowing, you may get advection out of or into the New York Bight. From this information, I would estimate the turnover time is a couple of weeks.

Question: Can you say whether the wave and wind energy over the study period were typical for the area? Were there more or fewer storms—bigger or smaller storms—than you would expect in the inner Bight?

J. Manning: A lot of these storms that occurred once or twice per month were about the same magnitude in terms of stress at the air–sea interface. I would not say that any one particular storm was anomalous. They were in the range of two to three dynes per square centimeter, typical of the inner Bight.

Response of the Hudson Shelf Valley Sewage Sludge-Sediment Reservoir to Cessation of Disposal at the 12-Mile Site

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ABSTRACT

One consequence of sewage sludge disposal at the 12-mile dumpsite in the New York Bight was accretion of a regional sediment-sludge reservoir. An assessment of its precessation extent and postcessation fate was undertaken to study the dynamics of this reservoir as a consequence of changing input. This assessment involved annual cruises to gauge the status of the reservoir through physical erosion experiments and measurement of chemical and microbiological contamination of representative stations. The key stations, chosen to represent the central stable reservoir, or near field, were distributed along the Hudson Shelf Valley from the northern extreme of the Christiaensen Basin to the southeasterly extreme of the Valley or entrance to the Hudson Canyon. A combination of sediment erodibility, *Clostridium perfringens* spore counts, and sterol chemistry were useful in portraying reservoir changes.

Assessment of the reservoir before cessation of disposal indicates major accumulation in the Christiaensen Basin as a watery, black sediment (i.e. sewage sludge), often covered with a *Beggiotoa* spp. network, growing to several centimeters above the sediment surface. However, some cruises indicated an apparent loss of this accumulation, which left behind a stable, anoxic silty sand. This pattern suggests that the area is a major deposition site for sludge, and periodic events resuspend that reservoir. It is hypothesized that down-valley turbidity plumes are the source of sludge deposition. Stations radiating out from the basin appear to have similar temporal patterns of accumulation but have smaller extremes in accumulation and loss. Deposition and temporary accumulation of silt-sized particles from other sources may continue along the valley after cessation of disposal.

It appears that continuous disposal resulted in a temporary sludge-dominated surface reservoir, primarily in upper valley depressions, that was highly susceptible to resuspension.

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*Formally with Science Applications International, Corporation.

ABSTRACT (continued)

An underlying anoxic sludge-sand reservoir was gradually winnowed and aerated over 1–2 years, following cessation, to an oxic silty sand. Disposal at the 6-mile dredged material (mud) dumpsite to the west of the sewage sludge dumpsite confuses the interpretation of survey results because of probable input of contaminants as well as bacterial spores from New York Harbor sediment. Deposition of contaminated fine sediment, from the mud dumpsite and the Hudson-Raritan Estuary, are likely to keep contaminants and biogeochemical rates at elevated levels in the uppermost surface sediments in the Christiaensen Basin. However, biological recovery is likely due to lower mass deposition and less severe hypoxic conditions, the postulated mechanism of damage of shallow water sludge disposal.

Introduction

For 64 years (1924–87), up to 8 million wet tons of sewage sludge was deposited annually at the 12-mile dumpsite (12-MDS) in the New York Bight (Fig. 1). The planned closure of the 12-MDS by the U.S. Environmental Protection Agency (EPA) by 31 December 1987 was seen by the National Marine Fisheries Service (NMFS), Environmental Processes Division (EPD), and the EPA's Atlantic Ecology Division at Narragansett (EPA-AED), as an opportunity to study the fate of sewage sludge-contaminated sediments in the area (EPD, 1988).

The EPA-AED has focused on those sediment-chemical reservoirs perceived to impact regional water quality and its biotic resources and on those processes responsible for changes in these reservoirs. In addition to local seabed problems such as degraded habitats and reduced productivity, the redistribution of particulate and dissolved components of these reservoirs may affect regional water and sediment quality. Reservoir solids, including vertically distributed microbes and low-solubility compounds (Fig. 2), represent the key exposures to marine biota, especially particle feeders. For instance, the transient fluid mud layer may contain the richest concentrations of organic carbon and pollutants, and the layer is commonly ingested by suspension and surface deposit feeders.

A bathymetric map of the New York Bight and limited lithologic studies (Freeland and Swift, 1978) suggest that fine-grained particles are likely to accumulate in the Hudson Shelf Valley (HSV), especially in the wider and deeper regions. Thus, it has been assumed and verified that sewage sludge disposed at the 12-MDS would settle and accumulate in the central Christiaensen Basin (CB) and portions of the upper HSV. Key indica-

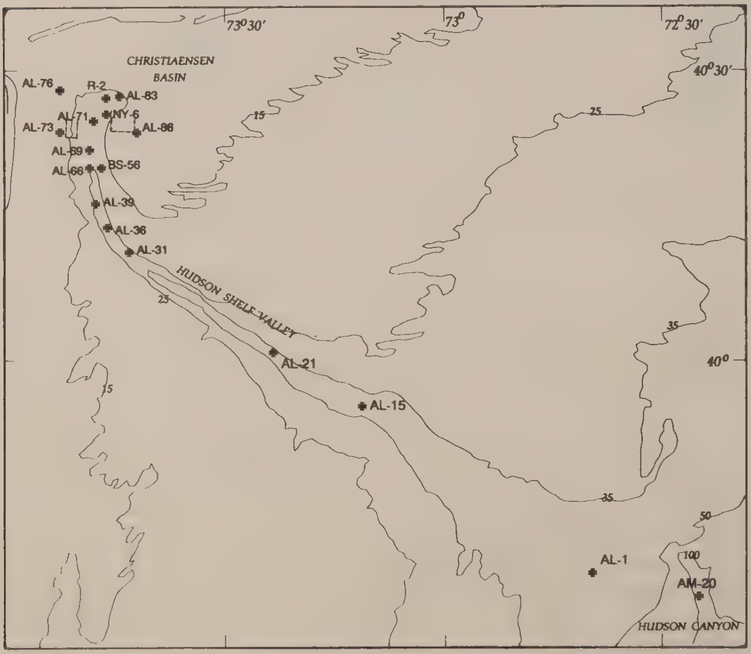

Figure 1

Map of New York Bight indicating sampling transect along axis of Christiaensen Basin-Hudson Shelf Valley-Hudson Canyon. The 12-mile dumpsite is located between NY-6 and AL-86 and the 6-mile dredged material (mud) dumpsite is located west of the AL-66/AL-39 transect (1–2 km).

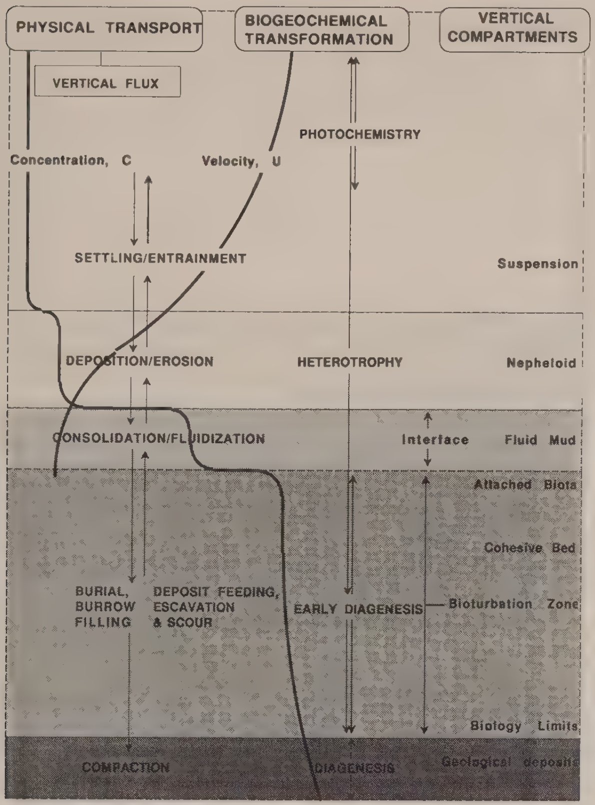
PHYSICAL TRANSPORT
BIOGEOCHEMICAL TRANSFORMATION
VERTICAL COMPARTMENTS
VERTICAL FLUX
PHOTOCHEMISTRY
Concentration, C
Velocity, U
SETTLING/ENTRAINMENT
Suspension
DEPOSITION/EROSION
HETEROTROPHY
Nepheloid
CONSOLIDATION/FLUIDIZATION
Interface
Fluid Mud
Attached Biota
Cohesive Bed
BURIAL, BURROW FILLING
DEPOSIT FEEDING, ESCAVATION & SCOUR
EARLY DIAGENESIS
Bioturbation Zone
Biology Limits
COMPACTION
DIAGENESIS
Geological deposits

Figure 2

Near-bottom particle transport and exposure compartments (C=suspended solids concentration, U = water velocity). Sewage sludge on the sediment surface behaves much as the fluid mud layer and, when mixed into the bioturbation layer, as anoxic sand-silt.

tors of sludge-contaminated sediment are the presence of *Clostridium perfringens* spores (Cabelli and Pedersen, 1982) and elevated levels of the fecal sterol coprostanol (Hatcher and McGillivary, 1979; Venkatesan and Kaplan, 1990). Understanding the distribution of these sludge indicators is critical to differentiating sludge contamination from that resulting from other sources of silt, such as the Hudson-Raritan Estuary plume and the dredged sediment disposal site (the 6-mile mud dumpsite), just west of the Basin (Fig. 1).

While the regulatory focus of sludge disposal at the 12-MDS is the contamination of sediment and shellfish with human pathogens, ecological effects may be the consequence of high deposition of fine particles and the biogeochemical consequences of the resulting reservoir of organic matter (i.e. high biological oxygen demand, low dissolved oxygen and sulfate, and the resulting mortality to postlarval and adult fauna). This hypothesis for ecological damage suggests that the impact of sludge disposal could be far greater than indicated by bulk chemical inventory. Consequently, recovery, recruitment, and growth of endemic benthic fauna may occur independently of any changes in bulk chemistry.

The objective of this study was to identify the sludge-sediment reservoir prior to and following cessation of disposal and to interpret the results in a transport context. Transport and fate of the sludge-sediment reservoir, following cessation of disposal, was interpreted from studies of sediment erodibility and its inventory of sludge markers (chemistry and microbial spores). A shift from unstable (e.g. highly erodible watery silt) to stable (e.g. sand-silt) could mean loss of a silty cap by resuspension, bedload transport of sand, or bioturbation mixing with underlying sand (Davis, 1993). Knowledge of chemical and microbial spore concentrations can add such new information, as would knowledge of recent storm events or dumping activity.

Methods

The study approach included assessment of sediment erosion potential and sediment inventory of sludge markers (*C. perfringens* spores, sterols, PCB's, and trace metals). Annual sampling cruises (1987–89) were made to determine temporal and spatial erodibility patterns of surface sediments and related patterns in sediment sludge markers. The final sampling cruise of the study area was made in June 1989, one and a half years following cessation of sludge disposal. The cruises assessed a sediment transect of the inner Bight, including the Christiaensen Basin (CB) and the Hudson Shelf Valley (Fig. 1). On each location along the transect, samples were taken for sediment chemistry (metals, PCBs and sterols), microbiology (*C. perfringens* spores), and physical erosion tests.

Sediment Erosion

Sediment erodibility was measured with shear tests performed with an oscillating perforated grid shearing device termed a Particle Entrainment Simulator (PES, Fig. 3) (Rouse, 1938; Davis et al, 1984; Tsai and Lick, 1986; Lavelle and Davis, 1987; Abdelrhman et al, in press). An undisturbed core taken from an undisturbed grab (no visible disturbance or resuspension) is subjected to a low shear level (2 dynes/cm^2) for up to an hour. Resuspension or erodibility is measured by the temporal accumulation of suspended solids over the sediment. The core is then similarly subjected to higher shear levels. A single test measures overlying suspended solids concentration with respect to time, permitting calculation of resuspension rates with respect to time

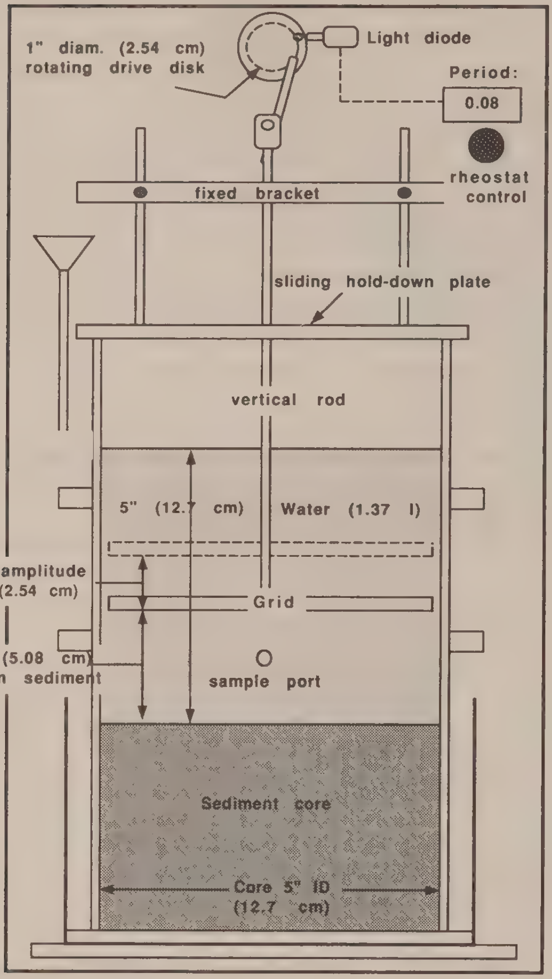

Figure 3
The Particle Entrainment Simulator, or PES. The grid is 0.25 in plexiglass, 11.0 cm diameter, with 36 1.20-cm holes drilled 1.50 cm apart.

and shear (Abdelrhman et al, in press). The PES is empirically calibrated with an annular flume (Fukuda and Lick, 1980). The experimental stress levels (a function of oscillation frequency) were calibrated over a range including 2, 3, 4 and 5 dynes/cm^2 (0.16, 0.12, 0.10 and 0.08 sec/cycle). Since most experiments reach steady state (resuspension = deposition), net resuspension with respect to shear can be estimated. A series of shear tests was performed and total flux in g/m^2 was measured at several sites, including the inner Bight and the HSV transect (Fig. 1). Three cores from each site were tested during each cruise. An example of three core experiments is shown in Figure 4.

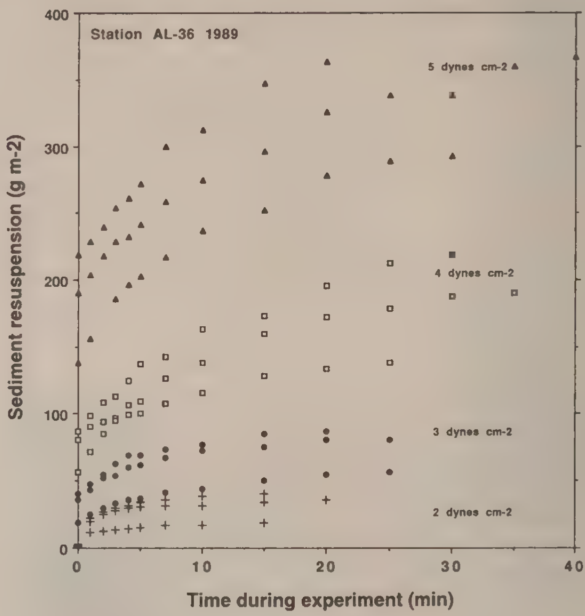

Figure 4
Experimental results of a Particle Entrainment Simulator (PES) test (AL-36-89: R/V *Albatross IV* Station 36, 1989). The test began at time zero and continued until suspended solids concentration were at or near steady state. Included in Figure are results from 3 replicate cores, each tested at 2, 3, 4, and 5 dynes/cm^2.

Sediment Chemistry

Sediment samples were extracted for PCB's using an acetonitrile/pentane extraction scheme described by Pruell et al. (1990). The internal standards included octachloronaphthalene for PCB's and 7,(5α-)cholesten–3ß-ol for sterol analysis. The concentrated extracts were fractionated using a 0.9 cm × 45 cm column which contained 11.5 g of BioSil A silicic acid (BioRad Laboratories) that had been fully activated, then 6.75% deactivated with water. The first fraction (f_1) was eluted with 45 ml of pentane. A second fraction of 35 ml of 30% methylene chloride in pentane was discarded, and a third fraction (f_3) was collected using 35 ml of 30% methanol in methylene chloride. The f_1 fraction was treated with activated copper powder to remove sulfur and then the f_1 and f_3 fractions were reduced in volume and exchanged to 1 ml of hexane.

The f1 fractions were analyzed for PCB's as Aroclor 1242 and Aroclor 1254 (Pruell et al. 1990). Each sample was injected (1 ml, splitless mode) into a Hewlett Packard 5890 gas chromatograph with an electron capture detector and a 30 m DB5 fused silica capillary column (J + W Scientific). Helium was the carrier gas at a flow rate of 1.5 ml/min; a flow of a 95:5 mixture of argon:methane to the detector was maintained at 35 ml/min. The oven temperature was held at 60°C for 1 min and then programmed from 60° to 315°C at 10°C/min. The injection port temperature was 275°C, and the detector was held at 325°C.

The f3 fractions were analyzed for sterols as above with the following modifications: the gas chromatograph was equipped with a flame ionization detector, and the oven temperature was held at 150°C for 1 min and then programmed from 60° to 315°C at 10°C/min. Measurement of total coprostanol (sum of coprostanol [5ß-cholestan–3ß-ol] and epicoprostanol [5ß-cholestan–3α-ol]) was made by comparing component/internal standard peak height ratios in chromatograms of the sample extracts to those of the quantitative sterol standards.

Trace metals were extracted from sediments by ultrasonic agitation with 2M HNO3, and the extracted metals were separated from the sediment residue by centrifugation. Inductively-coupled plasma emission spectrometry was used to quantify the extracted metals. The extraction procedure dissolves metals present in the sediment in most chemical forms but does not attack the sediment mineral matrix; consequently, the concentrations measured provided an estimate of the maximum environmentally available metal concentrations, rather than the absolute total concentrations of metals in the sediment.

Microbial Sludge Indicators

Clostridium perfringens spores in station grab samples were determined using a sonicate and settle extraction procedure (Emerson and Cabelli, 1982) coupled with a membrane filtration enumeration method (Bisson and Cabelli, 1979). Briefly, sediments were suspended in sterile distilled water, pulse-sonicated for 10 sec using a Branson Sonifier model 350 fitted with an 1/8 inch microtip at a 2-mil amplitude, and allowed to settle. Supernatants were examined by filtering through 0.45mm membranes, which were anaerobically incubated on MCP agar plates at 45°C for 20–24 hours. Colonies of *C. perfringens* on the surface of the membranes were confirmed by an in situ phosphatase assay. Spores of *C. perfringens* per gram wet weight of sediment are calculated from confirmed colony counts, and the densities shown are the means from triplicate determinations for each sample.

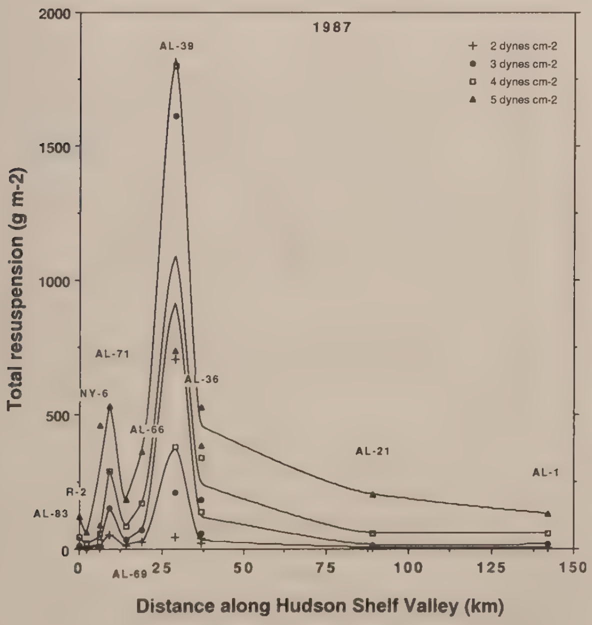

Figure 5

A summary of 1987 sediment erodibility in relation to Hudson Shelf Valley bathymetry. Each data set is labeled with the station prefix (AL-, NY-, R- or BS-), the station number (e.g. –39), and the cruise year (87, 88, or 89). The zero value is AL-83 (see Fig. 1).

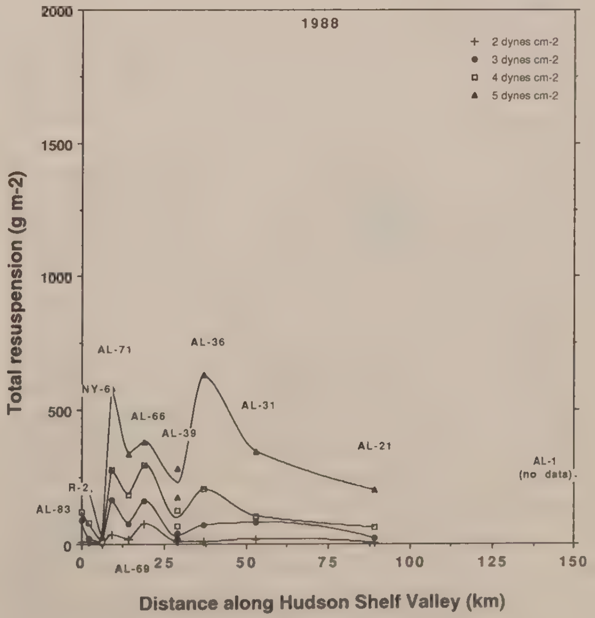

Figure 6

A summary of 1988 sediment erodibility in relation to Hudson Shelf Valley bathymetry (see Fig. 4 legend).

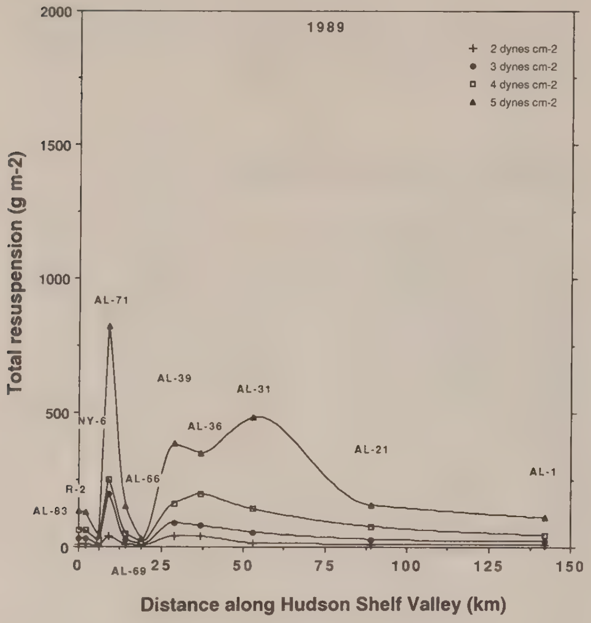

Figure 7

A summary of 1989 sediment erodibility in relation to Hudson Shelf Valley bathymetry (see Fig. 4 legend).

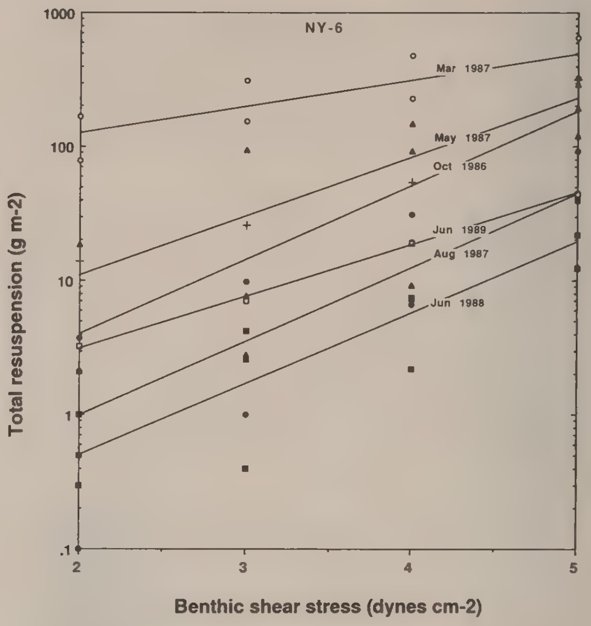

Figure 8

Summary erosion response at NY–6 (steady-state suspended solids concentration with respect to applied shear).

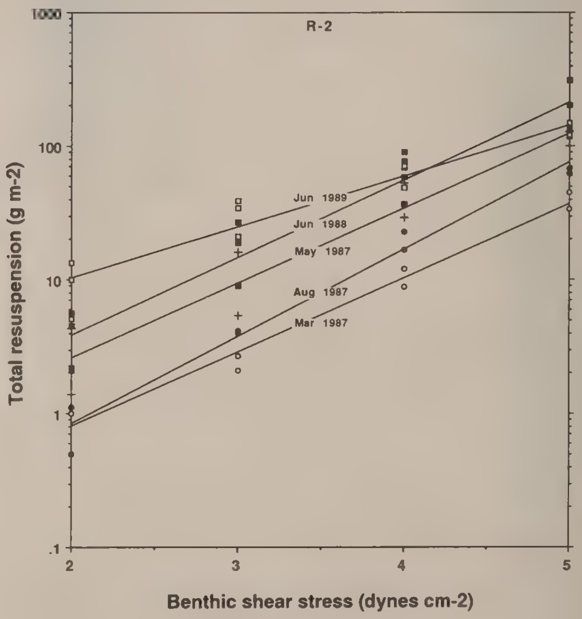

Figure 9

Summary erosion response at R-2 (steady state suspended solids concentration with respect to applied shear).

Results

Sediment Erosion

Results of each erosion test provide a measure of resuspended sediment (g/m^2) with respect to time (up to 60 min) and applied shear (2, 3, 4, and 5 dynes/cm^2; see Fig. 4). The slope of a best-fit asymptotic curve for each stress level is an experimental estimate of time-dependent resuspension rate (g/m^2·sec^2), and the final steady state value is an experimentally determined estimate of total resuspension at each shear stress level (Abdelrhman et al, in press). Sediment erodibility along the Christiaensen Basin and Hudson Shelf Valley Transect (Fig. 1) is summarized for representative stations along this transect for 1987 (Fig. 5; precessation) and for 1988 (Fig. 6) and 1989 (Fig. 7; postcessation). Each curve estimates the steady state value of the quantity of sediment resuspended as a function of experimentally applied shear stress.

Erodibility of inner New York Bight stations was highly variable in both space and time (Table 1). The sediment ranged from highly erodible (watery anoxic silt) to highly stable (medium to coarse sand and trace silt). Station erodibility along the CB-HSV transect varied according to depth and basin characteristics, valley width and distance from the 6 and 12-mile dumpsites and Harbor entrance. Upper CB stations (AL-83, R-2) are at the upper rim of a shallow depression and sediment varied between sand and sand-silt, with occasional silt (lowest erosion values in inner Bight) (Fig. 1). Mid CB stations (e.g. NY-6) varied between watery, anoxic silt,

Table 1
Replicate station sediment features and experimental erodibility (g/m^{-2} at steady state).

	STATIONS											
	NY-6				R-2				NY-11			
	2	3	4	5	2	3	4	5	2	3	4	5
						dynes/cm^2						
10/20/86												
Erodibility												
	2.1	2.8	9.2	121.0					(too porous to run)			
	18.6	93.2	149.0	294.0								
	2.1	7.7	93.0	192.0								
Sediment features												
	Anoxic sand-silt; large detritus (100–200 μ flakes); non-cohesive; *Cerianthiopsis*				Sand w/silt; shell frag.				Coarse, oxic sand, trace silt			
3/18/87												
	78.0	153.0	228.0	326.0	1.0	2.1	8.8	33.4	(too porous to run)			
	166.0	310.0	480.0	650.0	—	2.7	12.1	44.4				
	Watery anoxic silt w/*Beggiotoa* sp.								Coarse, oxic sand; trace silt			
5/12/87												
	13.8	25.6	54.2	314.0	1.4	5.4	29.4	99.5	(could not sample)			
					4.3	15.9	53.0	128.0				
8/18/87												
	3.7	9.7	19.2	45.2	0.5	4.0	16.8	62.6	1.2	7	33	82
	0.1	1.0	31.0	93.0	1.1	4.1	22.6	67.4				
	2.1	4.2	6.6	12.6								
	Anoxic silt, trace sand; large fibrous material detritous; hard sand-silt below 2 cm; dense *Beggiotoa*				Anoxic sand; trace silt; black flakes (100–300 μ) *Pherusa, Nepthys* dominant; *Cerianthiopsis*				Coarse, oxic sand; trace silt			
6/27/89												
	0.3	0.4	2.2	12.2	5.6	18.8	58.0	128.0	(too porous to run)			
	0.5	2.6	7.5	39.8	4.5	26.9	76.8	310.0				
	1.0	4.2	7.3	21.7	2.2	9.0	36.3	135.0				
					2.1	19.8	90.7	203.0				
	Aerobic sand; dense worm tubes above interface				Aerobic sand				Coarse, oxic sand; trace silt			
6/6/89												
	3.2	7.0	19.0	44.1	9.9	20.8	49.0	118.0	8.5	13	23	51
					5.1	34.2	69.0	148.0				
					13.3	39.0	70.0	120.0				
	Aerobic sand; very porous				Aerobic sand; anoxic sand-silt below 1 cm; [R] silt sand below 6 cm; mature *Nepthys*				Coarse, oxic sand; trace silt			

and semi-stable anoxic flakes (both essentially sewage sludge) in 1986–87 but shifted to anoxic sand-silt and, finally, oxic sand in 1988–89. Sediments at CB-HSV transition stations (e.g. AL-71), nearest to the 6-mile site, were generally silt-clay (unstable) throughout the study. Stations in the upper HSV (AL-69 and AL-66) consisted of semi-stable silt-clay throughout the study, suggesting slow accumulation of fines and minor resuspension events. The mid-upper HSV (AL-39, AL-36) was deeper, wider, and highly unstable in 1987 (watery anoxic silt and high spore levels) and was probably a sludge deposition area. Stabilization and reduced spore concentration after cessation (still silt) suggest that de-watering and cohesion followed reduced sludge deposition. Stability and sand content increased down valley (e.g. AL-31) and remained unchanged at AL-21 and AL-1.

Total resuspension with respect to shear stress for three inner Bight stations (NY-6, R-2, and NY-11, "replicate" stations) is summarized in Table 1 and Figures 8 and 9 (EPD, 1988; Pikanowski, 1995). Station NY-6 (Fig. 8), located in the Christiaensen Basin, was most contaminated by sewage sludge (as indicated by *C. perfringens* spores). Prior to cessation, NY-6 was periodically found to consist of mostly deposited sludge (25 cm grab depth). This condition (e.g. March 1987) was characterized by black, watery silt, covered with *Beggiotoa* spp., and high erodibility (Fig. 8; Table 1). On other occasions NY-6 was characterized by a stable and homogeneous anoxic sand-silt (e.g. September 1987); intermediate conditions were just as likely (e.g. October 1986). Temporal variability suggests strongly that NY-6 rapidly accumulated sludge during calm periods and, possibly, lost its silty cap reservoir during storm events. Since cessation (December 1987) NY-6 has gradually lost its anoxic silt, to become characterized by an aerobic sand with a minor silt fraction (1988) and a well-sorted sand with just a trace of silt (1989).

Station R-2 is located at the northern edge of a shallow depression in the Christiaensen Basin. The major pattern found at R-2 included (1) a precessation anoxic and stable sand-silt, (2) little evidence of accumulation of sludge-dominated sediments (anoxic watery silt), and (3) a gradual shift to oxic sand, consistent with high energy winnowing (Table 1, Fig. 9).

Sediment characteristics at stations not located on the CB-HSV transect were spatially patchy and suggest caution when interpreting any station. AL-86, located in the northeast section of the 12-MDS, consisted of aerobic coarse to medium sand throughout the study (Fig. 1). The only evidence for sludge was occasional deposits of anoxic sand-silt 2–5 cm beneath the surface. Station AL-76, located between the CB and the mouth of the Hudson-Raritan Estuary, varied between stable clean sand (precessation) and highly erodible silt-clay (postcessation). Station AL-73, located outside of the northwest corner of the 6-mile site, varied between semi-stable silt-clay (cohesive) and unstable silt-clay. Sampling (n=6) at the 6-mile site in 1989 gave erodibility results similar to the trend in AL-73 (Davis, unpublished data).

Sediment Chemistry

The decrease in levels of organic contaminants (as seen in May 1987) as a function of distance down-valley is consistent with a near coastal source of contaminants and subsequent transport and dispersal down-valley. The highly erodible sediments found at station AL-39,

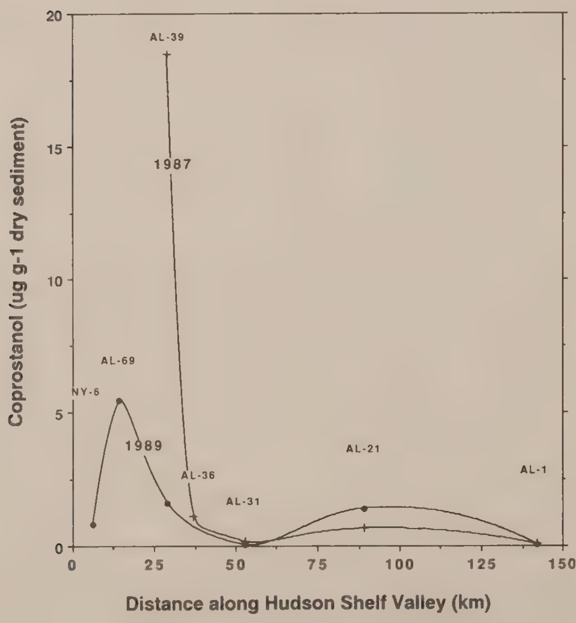

Figure 10
Total coprostanol concentration as a function of distance along the Hudson Shelf Valley.

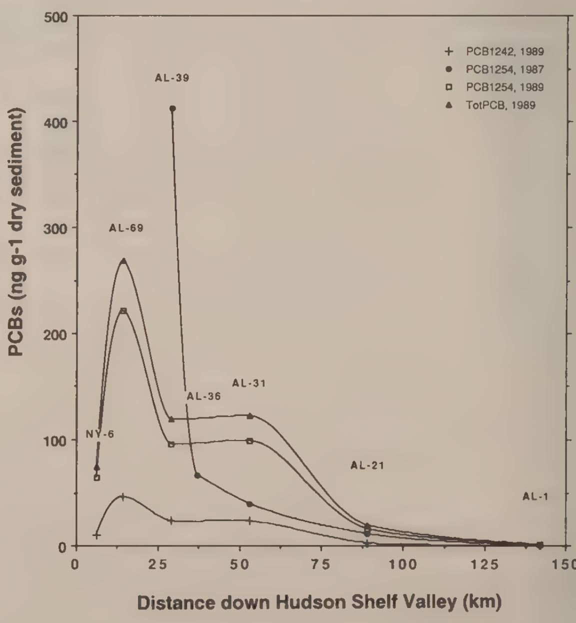

Figure 11
Total PCB concentration as a function of distance along the Hudson Shelf Valley.

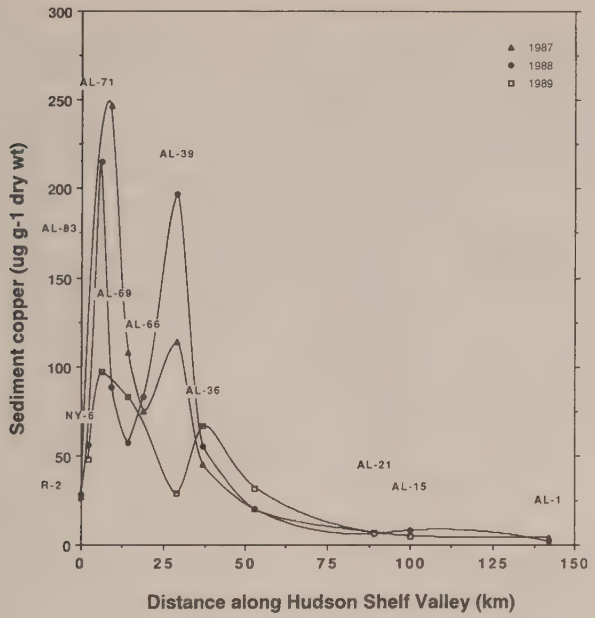

Figure 12

Summary mean copper concentration (1887, 1988, 1989) as a function of distance along the Hudson Shelf Valley (Warren Boothman, U.S. Environmental Protection Agency, Narragansett, RI, August 1992).

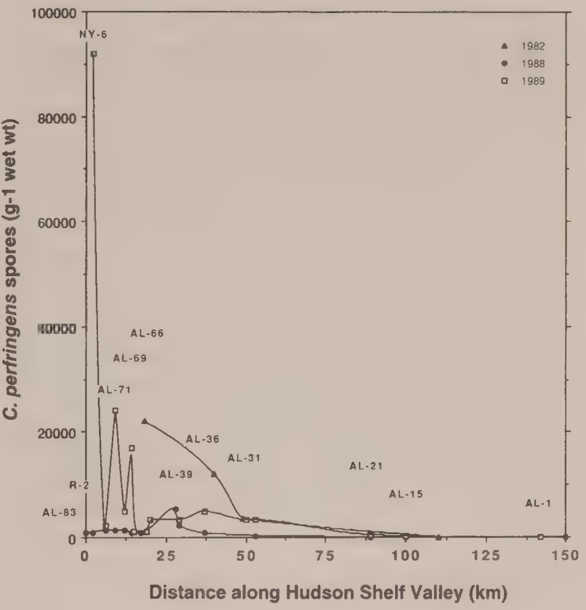

Figure 13

Trends in *Clostridium perfringens* spores along the CB-HSV transect prior to 1982, and after cessation in sludge disposal (1988, 1989).

where the valley deepens, had elevated total coprostanol concentrations ranging from 12 to 27 mg/gm (ppm) (Fig. 10). These values are consistent with those reported for sediments contaminated with sewage sludge (Hatcher and McGillivary, 1979) and suggest the presence of a significant amount of material of fecal origin. Concentrations of PCBs and coprostanol for 1987 and 1989 along the CB-HSV transect follow a similar pattern (Fig. 11).

Concentrations of trace metals in sediments from the Christiaensen Basin were highly variable, both between stations and among individual grabs at single stations (e.g. copper, Fig. 12). Mean concentrations of copper varied over two orders of magnitude within the basin area. At some stations, concentrations varied by as much as 100% between replicate grabs. The high degree of variability indicates a very patchy distribution of metals in the basin sediments. The highest concentrations of metals were generally found at stations AL-73 and AL-71, adjacent to the dredged material dump site (Boothman[1]). In addition to high contaminant metal concentrations, these sediments had substantially higher iron and manganese concentrations, indicating a high fine-grained material content typical of many dredged sediments. Somewhat elevated concentrations of metals were also found in some of the 1988 samples from the NY-6 station, possibly demonstrating a contribution from sewage sludge; such elevated levels were not, however, apparent in 1989 sediments. Likewise, sediments from station AL-86, also in close proximity to the 12-mile site, contained very low concentrations of metals. Values peaked near the 6-mile site (AL-71) and the wide depositional area (AL-39) but were reduced to less than half of 1987 levels by 1989 (e.g. copper; Fig. 12). Statistical analysis of relationships between metals concentrations and distance along the mid and lower Valley transect indicates that there was no significant difference among the years studied. Although some of the data from station NY-6 suggest the possibility that sludge from the 12-MDS could be contributing metals to the sediments of the HSV, the low metals concentrations found in sediments in the vicinity of the 12-MDS and the generally higher concentrations of metals in sediments from the dredged sediment disposal site (1–2 nm west of AL-66 and AL-39) implicate that site rather than the 12-MDS, as a likely source for the transport of metals to the HSV. Therefore, cessation of sludge dumping at the 12-MDS is unlikely to produce any significant change in metals concentrations in HSV sediments.

Microbiological Sludge Indicators

Because the 1987 sediment collection was lost, a 1982 survey (Cabelli and Pedersen, 1982) is used to represent the "steady state" dumping levels of *Clostridium perfringens* spores (Fig. 13). The 1982 survey measured decrease in spores with distance down the channel and represents steady state sediment contamination after years of sludge disposal. A survey approximately six months after complete cessation of dumping in December 1987 showed a complex distribution indicating (1) a 100-fold decrease in spores in the Christiaensen Basin and upper Hudson Shelf Valley, (2) a graded change to an order of magnitude decrease in the middle Valley, and (3) a gradual merging to 1982 background values at the base of the Valley. One year later (June 1989) the down-channel trend showed (1) a return to 1982 values in the CB, (2) wide fluctuations between 1982 highs and 1988 lows at the CB-HSV transition, (3) upper HSV values intermediate between 1982 and 1988 levels, and (4) a return to 1982 levels in the mid and lower HSV.

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[1] Boothman, Warren. U.S. Environmental Protection Agency, Narragansett, RI. Pers. comm., August 1992.

Discussion

Precessation

The observed patterns in erodibility in the Christiaensen Basin (e.g. NY-6) suggest that sewage sludge accumulated (possibly at a rate of 1 cm/day) during quiescent periods and was then resuspended by storm events. This temporary cap of black, watery silt (essentially sludge) apparently does not compact possibly owing to lack of time or bioturbation. When energy intensity and duration were sufficient, this watery cap was apparently resuspended, exposing an underlying base of anoxic sand-silt. This stable matrix may represent the long-term sludge-sediment reservoir that was regularly capped with dumped sewage sludge or with any source of silt-sized particles. It is hypothesized that down-valley deposition may originate from turbidity plumes moving from the Christiaensen Basin into the HSV, although no evidence exists.

Postcessation

Following cessation of sewage sludge disposal this cycle of accumulation and resuspension probably continues, but because the temporary silt cap is aerobic, the underlying base shifts toward aerobic sand-silt and, finally, to sand from combined resuspension and winnowing. A peak in measured erodibility shifted about 25 km down the HSV from 1987 to 1989, and measured erodibility declined in amplitude to about 30% of its 1987 values (Figs. 5–7). The key change may be a smaller quantity of silt-sized particle deposition, a less labile carbon reservoir, and thus an oxidizing state. The relic and future grain size of the CB-HSV may be sand (CB), with a transition to sandy silt in the upper-middle HSV, finally reversing toward sand in the lower HSV with deposition of fines consistent with suspended solids and quiet conditions.

Changes in the CB-HSV transect spore distribution after cessation suggest certain possibilities: (1) shallow stations in the Christiaensen Basin may lose their sludge markers rapidly due to greater resuspension activity, especially with the loss of the black, watery cap (a shift toward silty sand or sand support this contention) and (2) the upper HSV, while of similar depth, may have lost much less of its spore reservoir, as suggested by studies of Phoel et al. (1995), who have shown that this area has received more dredged sediment (New York-New Jersey harbor origins) and possesses a greater oxygen demand than expected; and (3) a depth-related down-valley spatial gradient in spore concentration may continue, but with a shifting baseline.

Clostridium perfringens spores appear to be one of the few reliable indicators available for assessing the "sanitary" quality of bottom sediments. Virtually all other bacterial indicators of contamination are subject to rapid density declines once deposited in the marine environment. The maximum extent of spore survival under such conditions is not known, however, viability generally persists for many months, even in frozen sediment. The unexpected increases seen for *C. perfringens* spores for the 1989 transect stations do not appear to be errors. The impact is seen to varying degrees at all stations along the HSV at a distance of more than 100 km. Analyses were carefully controlled, and because spores are known not to divide in such environments, these elevated levels are attributed to sewage sludge, either recently deposited or resuspended. An alternative source of spores could be from redistribution and resuspension of dredged sediment disposed at the 6-mile site. The U.S. Army Corps of Engineers records show a major increase in dumping of spoils in spring-fall 1989.

A variety of variables interact to result in transport-loss of sludge from area sediments including (1) the dramatic shift in input, (2) storm and tidal shear stress, and (3) a shift in biology (Davis and Means, 1989; Reid et al., 1995). However, chemical and spore data suggest that other inputs (e.g. 6-mile site) continue to contaminate surficial sediments through resuspension and subsequent deposition. Site processes (e.g. sludge disposal) and its related sediment condition (anoxic watery silt cap) are described as events that proceed sequentially, facilitating the next step of recovery (Fig. 14). It is speculated that the key impact of sludge disposal was a combination of high deposition of fines and hypoxic conditions related to its labile carbon. Rapid biological recovery (infaunal larval recruitment) is therefore predicted. The recovery of the chemical inventory, although beginning quite early, may be the last condition to recover.

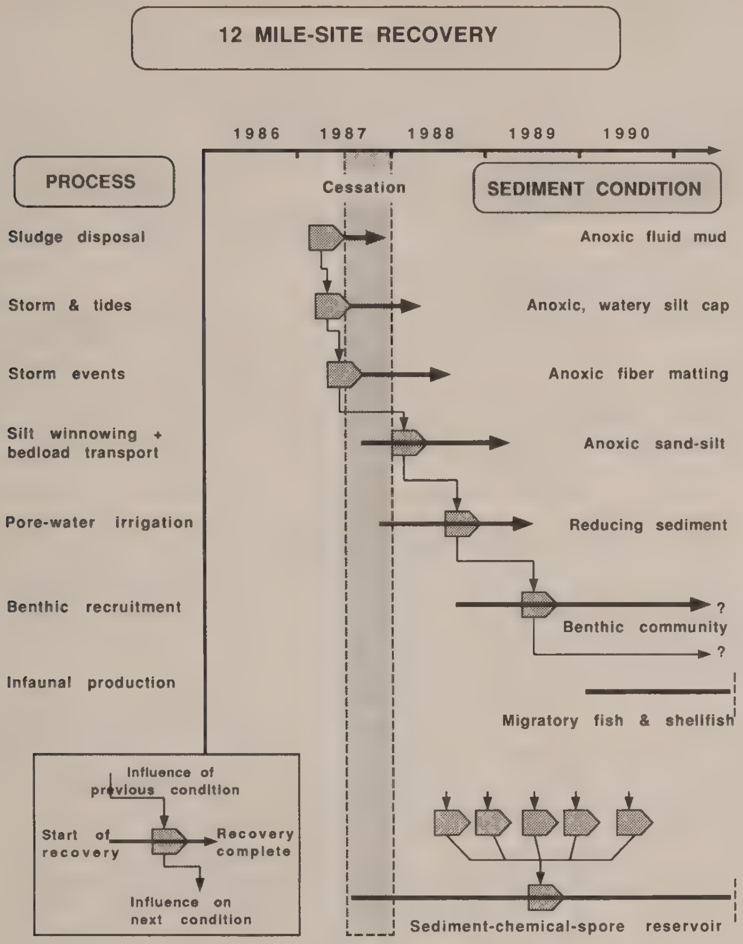
12 MILE-SITE RECOVERY
1986
1987
1988
1989
1990
PROCESS
Cessation
SEDIMENT CONDITION
Sludge disposal
Anoxic fluid mud
Storm & tides
Anoxic, watery silt cap
Storm events
Anoxic fiber matting
Silt winnowing + bedload transport
Anoxic sand-silt
Pore-water irrigation
Reducing sediment
Benthic recruitment
?
Benthic community
?
Infaunal production
Migratory fish & shellfish
Influence of previous condition
Start of recovery
Recovery complete
Influence on next condition
Sediment-chemical-spore reservoir

Figure 14
Possible trend in recovery of the 12-mile dumpsite. The first shift following cessation in sludge deposition is speculated to be resuspension of the anoxic watery silt top layer by storms and post-storm tides.

Conclusions

Erosion study results indicate that changes in sediment structure (grain size, porosity, and biological cohesion) have occurred in the HSV. The changes are consistent with resuspension and redistribution of contaminated particulates along the axis of the HSV. Long-term changes in the sludge-sediment reservoir will be the result of a complex interaction of various transport processes such as particle flux and in-sediment mixing (bioturbation). The following summary conclusions are justified at this time:

1. Evidence from measurements of erodibility, organic chemistry, and fecal spores indicate a regional sewage sludge-sediment reservoir with highest stability in wide, protected depressions of the CB-HSV transect. The reservoir rapidly declines with distance both away from and down the transect.
2. The surficial layer of the reservoir is highly transient because of major sludge accumulation followed by event-related resuspension while deeper sediment is a stable sand-sludge mixture.
3. Since cessation of sludge disposal, the sediment regimes have shifted from reducing to oxidizing conditions, from silt-dominated to sand-dominated sediments, and have shown a reduction in indicators of fecal contamination (sterols and spores).
4. Increased dumping of New York harbor sediments at the 6-mile site appears to have resulted in a reversal in the declining trend in *C. perfringens* spores.
5. The experiments performed in this study suggest that existing seabed maps of lithologic features (Freeland and Swift, 1973) may be transformed to express resuspension potential or shear stress patterns.

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Recent Sediment and Contaminant Distributions in the Hudson Shelf Valley*

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ABSTRACT

Net sediment accumulation rates were determined in the vicinity of the nearshore dumpsites (Area 1) through measurements of radionuclide tracers in sediment core sections. The highest rates of accumulation (a few cm/y) were found along the axis of the Hudson Shelf Valley (HSV) up to several kilometers down valley from the dredge spoil and former sewage-sludge disposal sites. Observed levels of PCB's, chlordane, and DDT-derived compounds in surface sediment samples were highest in cores characterized by fine-particle deposition. Contaminant levels were much lower in samples of recent sediment from the HSV at mid-shelf (Area 2) and on the continental slope (Area 3), reflecting minimal influence of the nearshore disposals. Analyses of samples from the continental slope indicated that DDT contamination of recent sediments from Areas 2 and 3 reflected regional inputs of unaltered DDT derived from atmospheric deposition. Evidence of the atmospheric DDT signal, a high ratio of pp'-DDT to pp'-DDD, was also found in some surface suspended particle samples from Area 1, near the dumpsites. This indicates that analyses of DDT-derived compounds can provide useful information on the behavior and fate of particle-associated contaminants in the nearshore environment. A few samples were analyzed for 2,3,7,8-TCDD. The highest concentrations, a few hundred parts per trillion, were found in the HSV between the sludge and dredge spoil dumpsites and were probably derived from disposal of material dredged from the lower Passaic River and Newark Bay, the site of a major incident of 2,3,7,8-TCDD contamination.

Introduction

This study was designed to provide information on the distribution, fate, and sources of particle-associated pollutants in the vicinity of the nearshore dumpsites through the application of a multi-tracer approach. Both sediment core (Table 1) and suspended particle samples were analyzed. The tracers employed included radionuclides (Cs-137 and K-40) and persistent chlorinated hydrocarbons (PCB's, DDT-derived compounds, chlordane, and dioxin).

The study area (Fig. 1) receives multiple, potentially significant inputs of contaminants from a number of sources. These include discharge from the Hudson,

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Table 1
Sediment core locations.

Station	Water Depth (m)	Latitude	Longitude
Area 1			
NY5	35	40°24.9'N	73°47.6'W
NY6	30	40°25.0'N	73°45.6'W
A41	40	40°23.4'N	73°46.8'W
A56	55	40°19.0'N	73°47.4'W
HSV1	60	40°16.9'N	73°47.2'W
HSV2	60	40°15.4'N	73°46.8'W
HSV3	60	40°13.9'N	73°45.9'W
HSV4	50	40°12.0'N	73°45.4'W
HSV5	60	40°10.3'N	73°42.0'W
Area 2			
HSV6	80	40°02.3'N	73°26.4'W
HSV7	80	40°00.9'N	73°20.4'W
Area 3			
HSV8	480	39°29.7'N	72°20.1'W
HSV9	980	39°28.4'N	72°15.2'W
EN-123 BC-E6	1640	39°45.8'N	70°55.4'W

Raritan, Passaic, and Hackensack Rivers (Hudson-Raritan discharge), disposal of dredged material from the lower Hudson River, Newark Bay, Raritan Bay, and contiguous waters (the New York Harbor complex) and past disposal of sewage sludge from New York City and New Jersey (Stanford and Young, 1988). Construction and demolition wastes, often referred to as cellar dirt, have also been dumped in the study area. Loadings of contaminants associated with this activity are believed to be negligible compared with the other sources (Stanford and Young, 1988). For many contaminants, the sources are closely related. For example, lower levels of DDT in sewage, resulting from the ban on domestic use in 1972, would lower not only the levels in sewage sludge but also levels in wastewater discharge, a major source of contaminants to the New York Harbor complex (Mueller et al., 1982). This would result in lower levels of DDT in the Hudson-Raritan discharge and in the recent sediments dredged from the harbor and disposed of in the study area. Such considerations make it difficult to quantify the relative importance of these sources on the basis of contaminant analyses. Only in the case of DDT-derived compounds and dioxins is some distinction of sources possible. Evidence for an atmospheric source of DDT-derived compounds is presented below and dioxin contamination in the study area is probably related to an industrial discharge in the lower Passaic River that significantly contaminated sediments in Newark Bay (Bopp and Simpson[1]). A specific indicator of sewage-sludge-derived material in sediments of the HSV, the abundance of *Clostridium perfringens* spores, is discussed in detail elsewhere in this volume (Davis et al., 1995; O'Reilly et al., 1995).

Accumulation of particle-reactive contaminants would be greatest in areas of recent deposition of fine-grained sediment. Such areas are relatively uncommon on the continental shelf off the northeast coast of the United States where sandy surface sediments are the rule (Schlee, 1973; Biscaye et al., 1978). These considerations led us to focus our sediment core sampling on specific areas along the axis of the HSV where the surface sediments were known to be fine-grained.

It is common to establish the "recent" nature of fine-grained sediment accumulation through the use of radionuclide tracers with known sources and input histories (Alderton, 1985). Cs-137 has been shown to be a useful particle tracer in freshwater, estuarine, and coastal systems (Olsen et al., 1989; Mulholland et al., 1992). The major sources of Cs-137 to natural waters include global fallout derived from atmospheric testing of large nuclear weapons and effluent from nuclear reactors. Sediment cores from areas of continuous, rapid net particle accumulation (on the order of 1 cm/y or more) can often be dated by their depth profile of Cs-137 activity (Olsen et al., 1981). In profiles that reflect the delivery of fallout-derived Cs-137 to the sediment column, the initial appearance of that radionuclide corresponds to approximately 1952, the beginning of large-scale atmospheric nuclear weapons testing. Maximum concentrations of Cs-137 would correspond to the fallout maximum of the mid 1960's (Hardy, 1977). Such detailed chronological information cannot be derived from sediment cores from the vicinity of the nearshore dumpsites because of the mixing inherent in the process of dredged material disposal, a major source of fine-grained particles to this area. For this study, the presence of Cs-137 in a sediment core section has been interpreted simply as an indicator of a post–1950 component in that sample. In cores with high net accumulation rates of fine particles, the Cs-137 penetration depth, i.e. the depth to which Cs-137 activity can be detected, would correspond to the amount of deposition since the early 1950's.

In areas of lower net sediment accumulation, it becomes more likely that biological mixing of the sediments will be an important factor in determining the

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[1] Bopp, R. F., and H. J. Simpson. 1991. Sediment sampling and radionuclide and chlorinated hydrocarbon analysis in Newark Bay and the Hackensack and Passaic Rivers. State of New Jersey Department of Environmental Protection, Final report for contract P24096.

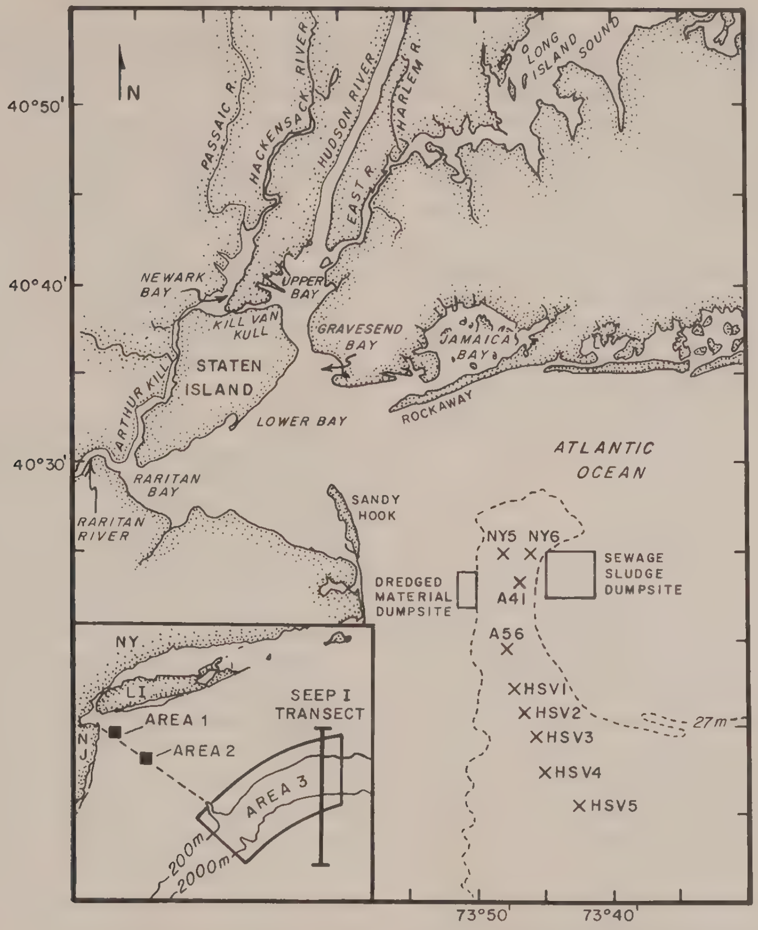
N
40°50'
40°40'
40°30'
PASSAIC R.
HACKENSACK RIVER
HUDSON RIVER
HARLEM R.
EAST R.
LONG ISLAND SOUND
NEWARK BAY
UPPER BAY
KILL VAN KULL
GRAVESEND BAY
JAMAICA BAY
STATEN ISLAND
ARTHUR KILL
LOWER BAY
ROCKAWAY
ATLANTIC OCEAN
RARITAN BAY
RARITAN RIVER
SANDY HOOK
NY5
NY6
A41
A56
SEWAGE SLUDGE DUMPSITE
DREDGED MATERIAL DUMPSITE
HSV1
HSV2
HSV3
HSV4
HSV5
27m
NY.
L.I.
N.J.
AREA 1
AREA 2
AREA 3
SEEP I TRANSECT
200m
2000m
73°50'
73°40'

Figure 1

Location of Area 1 sampling sites. The axis of the Hudson Shelf Valley falls within the area bordered by the 27 m contour line. The cellar dirt dumpsite is located between the dredged spoil and former sewage sludge dumpsites. Approximate locations of sediment coring sites in Area 1 are indicated by the x's. The insert shows the general locations of Areas 2 and 3. Area 2 includes coring sites HSV6 and HSV7. Sites HSV8 and HSV9 are in Area 3, on the axis of the Hudson Canyon. Coring site EN-123, BC-E6, and the sediment trap moorings were located along the SEEP-I transect. Latitudes and longitudes of all coring sites are listed in Table 1.

Cs-137 penetration depth, (Goldberg and Koide, 1962; Guinasso and Schink, 1975; Robbins and Edgington, 1975; Olsen et al., 1981). Biological mixing will produce lower Cs-137 activities by the dilution of recent deposition with sediments that accumulated prior to 1952. Similarly, lower concentrations of particle-associated contaminants would result.

Sediments with higher proportions of sand-sized particles would also be expected to have lower Cs-137 activities and particle-associated contaminant levels than their finer-grained analogs (O'Connor, 1988). Two processes that could increase the fraction of coarser particles in a sediment sample are post-depositional winnowing of fine particles and biological mixing of recent fine-particle deposition into older coarser sediments. In either case, this sediment type would be characterized by lower activities of K-40, a natural radioactive tracer of the distribution of stable potassium. Since potassium is enriched in clay minerals relative to quartz sands, the activity of K-40 in a sediment sample provides a useful first-order indication of grain-size distribution.

Based on the preceding discussion, sediment cores were collected to determine the depth of Cs-137 penetration, average Cs-137 activity, average K-40 activity and contaminant concentrations. The radionuclide data provided an indication of the extent and degree of influence of Hudson-Raritan discharge and sewage sludge and dredged spoil disposal on contaminant levels in recent sediments of the study area.

Samples of suspended particles were collected to characterize Hudson-Raritan discharge and to investigate the possibility of particle resuspension from the disposal sites. The ratio of pp'-DDT to pp'-DDD was used as a tracer of the atmospheric input of DDT-derived compounds. Such inputs are characterized by a high ratio of pp'-DDT to pp'-DDD (Rapaport et al., 1985). pp'-DDT is the dominant isomer in technical DDT and is degraded to pp'-DDD under anaerobic conditions that can be found in soils, sediments, and sewage treatment plants.

Methods

Sediment samples were collected by gravity or piston coring. Cores were sectioned at 1 to 8 cm intervals and dried at 35°C under a flow of air filtered through Florisil to minimize atmospheric contamination. Core sections were ground with a mortar and pestle and stored in airtight aluminum cans. Radionuclide analysis was accomplished by non-destructive gamma counting, employing lithium-drifted germanium and intrinsic germanium detectors and multi-channel analyzers.

Suspended particle samples were collected with the large volume in situ filtering system (LVFS) developed by James Bishop of the Lamont-Doherty Geological Observatory (Bishop et al., 1985). The filters used were 25.4-cm-diameter quartz fiber (Whatman QMA) that were pre-combusted at 400°C for at least 42 hours. Sediment trap samples were collected in traps deployed on moorings, described in Biscaye et al. (1988), as part of the first Shelf Edge Exchange Processes (SEEP-I) experiment.

Subsamples of sediment core sections and individual filters were soxhlet extracted, cleaned, and analyzed for PCB's, DDT-derived compounds, and chlordane by electron capture gas chromatography (Bopp et al., 1981 and 1982). Dioxin analyses employed isotopic spikes, chromatographic clean up, and high resolution gas chromatography/high resolution mass spectrometry as described previously (Tong et al., 1990). All contaminant concentrations and radionuclide activities are reported on a dry weight basis.

Sediment cores were obtained from three general areas (Table 1; Fig. 1). The first (Area 1) was in the immediate vicinity of the dumpsites. Specific coring locations were chosen on the basis of surface sediment textures determined during the Northeast Fisheries Center's 12 mile dumpsite Broadscale Survey (EPD [Environmental Processes Division], 1988). Our desire to obtain fine-grained sediments led to the collection of cores at stations NY5, NY6, A41, and A56 (Fig. 1). Field observations of sediment types at these sites made during the 12-mile dumpsite surveys ranged from very soft soupy mud (A41 and A56), to medium soft sludgy mud (NY6), to compact mud (NY5) (Reid et al., 1989). Our cores from station A56, in the axis of the HSV, about 6 km south of the dumpsites, were dominated by a fairly homogeneous fine-grained mud throughout their entire length. This observation prompted our collection of cores at five additional sites along the axis of the valley up to about 18 km down-valley from A56 (HSV1 - HSV5).

The second coring area (Area 2) was located about 45 km down-valley from A56 (Table 1; Fig. 1). It was chosen on the basis of grain-size maps prepared by one of the authors (Biscaye) which indicated that it was one of the few mid-shelf areas in this general region with fine-grained surface sediments (40–60% <63 mm). Cores were collected from stations HSV6 and HSV7 from within this surface textural regime.

Sediment cores were also collected from the continental slope (Area 3; Table 1; Fig. 1), an area known for its fine-grained surface sediments. Coring sites HSV8 and HSV9 were on the axis of the Hudson Canyon about 150 km down-valley from A56. In addition, results from core EN-123–BC-E6 will also be discussed. This core was collected on the continental slope about 200 km northeast of the Hudson Canyon as part of the SEEP program. It has been described in detail in a previous publication (Anderson et al., 1988).

Results

Radionuclide Activities in Sediment Cores

Available grain-size information supports our assumption that K-40 levels in our samples provide a first-order indication of particle-size distributions (Table 2; Appendix Table 1). Average K-40 levels were highest (>18 pCi/g) in Area 3 cores from the Hudson Canyon (HSV8, HSV9) where previous studies found that 80–100% of the surface sediments were <63 mm (Schlee, 1973; Biscaye et al., 1978). Much lower average K-40 levels (<12 pCi/g) were found in cores from Area 2 (HSV6, HSV7) where 40–60% of the surface sediments were <63 mm. Additional measurements relating grain size to K-40 activities in sediments from our study area would be quite useful.

Cs-137 penetration depths were greatest in Area 1 along the axis of the shelf valley, reaching a maximum of 2 meters in a piston core from site A56 (Table 2). The average Cs-137 and K-40 values at sites A41, A56,

Table 2

Radionuclide data on sediment cores. (Cores are listed by location starting from the head of the Hudson Shelf Valley [NY5, NY6] to the Hudson Canyon [HSV8, HSV9]. See Fig. 1 for location; Appendix Table 1 for all core section values.)

Core[1]		Date of collection	Cs-137[2] penetration (cm)	Average Cs-137[3] (pCi/kg)	Average K-40 (pCi/g)
AREA 1					
NY5	#1	10/30/87	16	54	10.7
NY6	#1	10/30/87	12	38	8.5
A41	#1	10/30/87	20	290	13.9
A56	#3	09/01/87	>47[4]	240	13.4
	#4	09/01/87	>48	240	13.4
	#5	09/26/88	>40	260	14.3
	#6	10/04/89	>40	220	14.6
	P1	09/26/88	>80	240	14.7
	P2	10/04/89	200	220	14.1
HSV1	#1	06/27/88	>19	330	15.7
	#2	06/27/88	>16	4	16.3
HSV2	#2	06/27/88	>21	140	13.3
	#3	10/04/89	20	59	11.7
HSV3	#1	06/27/88	>21	110	12.4
HSV4	#2	11/17/88	20	50	11.1
HSV5	#1	11/17/88	24	44	10.4
AREA 2					
HSV6	#1	11/17/88	6	37	11.0
	#2	11/17/88	6	25	11.5
HSV7	#1	11/17/88	20	27	10.2
AREA 3					
HSV8	#1	11/17/88	6	85	18.7
HSV9	#1	11/17/88	0	0	18.5
EN-123 BC-E6		10/25/84	4[5]	95	14.5
N.Y. HARBOR					
–1.7		09/03/79	40	500	16.8
NB 13		08/21/85	32	260	13.5
RB 17		05/20/80	>29	140	11.7

[1] P indicates piston cores. All others were gravity cores except EN-123 BC-E6 which was a subcore of a box core.

[2] Cs-137 penetration is defined as the bottom of the deepest core section that contained Cs-137 at a level at least two standard deviations greater than zero. Using this two sigma statistical counting error criterion, our minimum detectable Cs-137 activities were typically 15 to 30 pCi/kg.

[3] Cs-137 and K-40 activities are mass weighted averages from the top section of the cores to the Cs-137 penetration depths. The only exception is core HSV9 #1 which had no detectable Cs-137. The K-40 activity reported for this core is the mass weighted average of the 0–1, 1–2 and 2–3 cm sections.

[4] > values indicate that Cs-137 was detected in the bottom section of the core except in the case of core A56 P1 where radionuclide profiles indicated that a section of the core top was missing. While this is not uncommon in piston cores, our gravity cores all had undisturbed sediment-water interfaces.

[5] Based on analyses of DDT-derived compounds, mixing in this core can be detected to a depth of about 7 cm, however, the Cs-137 activity was within two standard deviations of zero by the 4–5 cm section (24 pCi/kg with a standard deviation of 17 pCi/kg).

and HSV1 are similar to those found in cores from areas of fine-grained sediment deposition in the New York Harbor complex (Table 2). These factors indicate that the sedimentary regime is characterized by fine particle accumulation. The rapid rate of recent net fine-particle accumulation at A56 was confirmed by the presence of significant Be–7 in the 0–2 cm sections of several of the cores from this site (Appendix Table 1). Be–7 is a cosmic-ray produced radionuclide with a half-life of only 53 days, which means that its detection in a sediment core section is an indicator of deposition within about a year of core collection.

Cores from the remaining sites in Area 1, those down-valley from HSV1 and those from sites NY5 and NY6, generally have lower average values of both Cs-137 and K-40 (Table 2). This could be the result of post-depositional winnowing of fine-grained sediments or biological mixing of recent, fine-grained deposition into older, coarser sediments. Lower average Cs-137 and K-40 levels were also seen in the cores from the mid-shelf, Area 2. The consistent variation of average Cs-137 levels with average K-40 levels throughout Areas 1 and 2 (Fig. 2) requires the blending of only two components; recent fine particles (enriched in Cs-137 and K-40) and sands (with low levels of Cs-137 and K-40).

Cores from Area 3, the continental slope, have much higher ratios of average K-40 to Cs-137 activities (Table 2) deviate significantly from the trend defined by cores from Areas 1 and 2 and those from New York Harbor (Fig. 2). At the site of core EN-123–BC-E6, the sedimentary regime has been described (Anderson et al., 1988) as consisting of low rates (on the order of one centimeter per hundred years) of recent fine particle accumulation. Profiles with depth of natural and fallout-derived radionuclides in the sediments were interpreted as resulting from biological mixing which typically occurs to a depth of several centimeters. Over this interval, the recent fines were mixed with much older, fine-grained sediments, diluting the average Cs-137 activity while retaining the high levels of K-40. Additionally, the relatively low efficiency of Cs-137 sorption onto particles from ocean water and the possible desorption of Cs-137 from particles into ocean water may contribute to the relatively high K-40 to Cs-137 activity ratios in Area 3 cores. Mulholland and Olsen (1992) have proposed that the former was responsible for lower Cs-137 activities observed on marine particles compared to particles derived from a riverine drainage basin. They also concluded that desorption of Cs-137 from the riverine particles upon transport into seawater was not significant because most of the Cs-137 was at nonexchangeable interlayer sites on clay minerals.

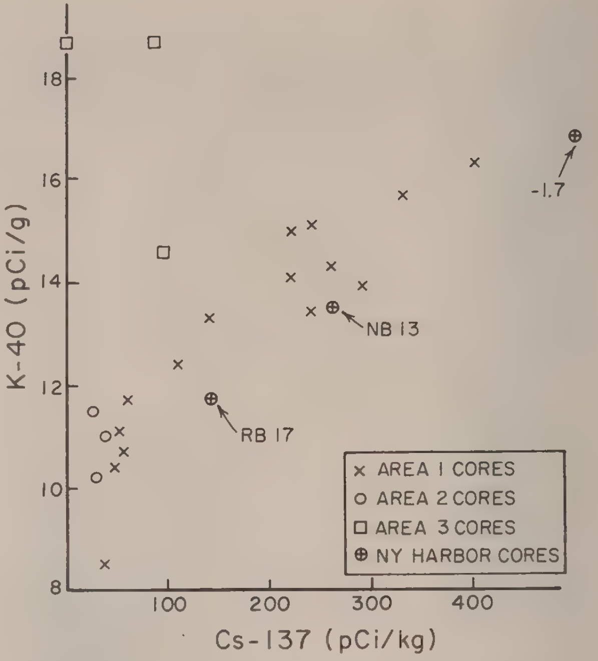

Figure 2

Average K-40 versus average Cs-137 activities in sediment cores. Area 3 cores have the highest ratios of K-40 to Cs-137 activity. Statistical counting errors for K-40 levels in individual core sections are on the order of ±1 pCi/g (see Appendix Table 1). This limits any attempt to distinguish between Area 1 and 2 cores and New York Harbor cores on the basis of this data.

Data from cores collected at three sites in the New York Harbor complex, chosen on the basis of their Cs-137 profiles, indicated continuous deposition since the early 1950's (Fig. 2; Bopp and Simpson, 1989; Bopp and Simpson[2]; Bopp and Simpson[1]). Core –1.7 was collected in 1979 from the Hudson River, 1.7 statute miles down channel from the southern tip of Manhattan Island. NB 13 was taken in Newark Bay in 1985, and RB 17 was from Raritan Bay in 1980. The three cores plot along the trend defined by cores from Areas 1 and 2, suggesting a Hudson-Raritan influence (either dredged spoil disposal or river discharge) on Area 1 and 2 sediments. The rapid rate of net sediment accumulation over the past three decades at some of our Area 1 sampling sites (for example, the two meters of recent sediment observed in piston core 2 at site A56; Table 2), would be consistent with dredge spoil translocation. The large volume of dredged material disposed annually (on the order of four million m^3/y; Gross, 1976, Stanford and Young, 1988) within 6 km of A56 suggest it is the likely source of sediment. We would have difficulty arguing that such high net accumulation rates could be supported by natural discharge of particles from the Hudson-Raritan Estuary.

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[2] Bopp, R. F., and H. J. Simpson. 1984. Persistent chlorinated hydrocarbon contaminants in the New York Harbor complex. Hudson River Foundation, First year final report for contract HUD 1183-A38.

Chlorinated Hydrocarbon Contaminants

Several core top sections were analyzed for chlorinated hydrocarbon contaminants including PCB's, DDT-derived compounds, and chlordane (Table 3). As expected, the highest levels of contamination were found in the cores from Area 1 that had the highest average Cs-137 and K-40 levels (sites A41, A56, and HSV1). The sections of the core from site A41 that contained measurable Cs-137 were composited and analyzed. This sample (A41, 0–20 cm, Table 2) should represent an average of the sediment deposited at this site since the early 1950's. Total PCB, pp'-DDD, and γ-chlordane levels were significantly higher than in the 0–2 cm section from the same core. This core section would represent the most recent deposition at this site. These results indicate a recent decline in the concentrations of these contaminants in sediments being deposited in Area 1. Similar improvements have been reported for sediments deposited in the lower Hudson River and were interpreted as the result of domestic restrictions on the use of all of these substances during the 1970's and, for PCB's, as a result of reduced transport from the upper Hudson (Bopp and Simpson, 1989). These changes would also decrease the concentrations of these contaminants in dredged material, sewage sludge, and Hudson-Raritan discharge, and would explain the re-

Table 3

Contaminant concentrations in surface sediment core sections from areas 1, 2, and 3 (see Fig. 1 for location).

Core		Depth (cm)	γ-chlordane[1] (ppb)	pp'-DDD (ppb)	pp'-DDT (ppb)	pp'-DDT / pp'-DDD	PCB's[2] (ppb)
AREA 1							
NY5	#1	0–2	7	22	5.0	0.23	(400)[3]
NY6	#1	0–2	14	18	29	1.61	(400)[3]
A41	#1	0–2	19	38	10	0.26	(950)[3]
		0–20[4]	24	117	75	0.64	1650
A56	#3	0–2	9	25	19	0.76	820
	#4	0–2	13	25	5.4	0.22	(750)[3]
HSV1	#1	0–2	14	115	41	0.36	820
HSV2	#2	0–2	4	14	25	1.79	290
HSV3	#1	0–2	3	7	1.4	0.20	200
HSV4	#2	0–2	0.3	1.1	<0.2	<0.2	50
HSV5	#1	0–2	0.9	2.0	<0.4	<0.2	60
AREA 2							
HSV6	#1	0–2	<0.1	0.5	<0.2	<0.4	<10
HSV7	#1	0–2	<0.1	0.5	<0.2	<0.4	<10
AREA 3							
HSV8	#1	0–1	<0.1	1.3	<0.2	<0.2	<10
EN 123 BC-E6[5]		0–1	<0.1	0.11	2.95	27	<5

[1] Technical chlordane is a mixture containing over 100 compounds. γ-chlordane, one of two major isomers, comprises about 15% of technical formulations (Dearth and Hites 1991).

[2] Based on quantification of 22 components resolved by packed-column gas chromatography (tri- through decachlorobiphenyl) (Webb and McCall 1973).

[3] Chromatograms contained significant interference from unidentified peaks. Concentration estimates are given in parentheses.

[4] This sample is a mass weighted average of the sections of this core that contained detectable activity of Cs-137. It represents an average of the sediments deposited at this site since the early 1950's. The 0–2 cm sample from this core should represent more recent deposition.

[5] The profiles of DDT-derived compounds with depth in this core were reported and modeled by Anderson et al. (1988).

cent decline in chlorinated hydrocarbon levels in Area 1 sediments that was indicated by our results from site A41.

Down valley from the sites of highest surface sediment contamination there is a decrease in contaminant levels at Area 1, sites HSV2 through HSV5, that generally follows the drop in the average Cs-137 and K-40 levels (Table 3). This indicates a decreasing contribution of contaminated fine-particles.

In the uppermost section of cores from the mid-shelf sites of Area 2, total PCB's and γ-chlordane were not detected at 10 and 0.1 ppb levels respectively. This decrease in contaminant concentrations relative to levels observed at sites HSV4 and HSV5 is greater than would have been predicted from the average Cs-137 or K-40 activities in the cores. For example, the decrease in average Cs-137 levels is at most a factor of two (Table 2), whereas the total PCB level drops by at least a factor of five and γ-chlordane drops by at least a factor of three (Table 3). This indicates that the fine-grained particles accumulating in Area 2 are significantly less contaminated with chlorinated hydrocarbons than those accumulating in Area 1.

Sediment core data from Area 3, the continental slope, show similar evidence of a less contaminated, recent, fine-grained component, i.e. undetectable levels of PCB's and γ-chlordane in samples containing Cs-137 (Table 3). Interpretation of the levels of DDT-derived compounds in sediments from Areas 2 and 3 provides a means of characterizing this component. In areas of low net accumulation, but significant biological mixing, the Cs-137 activity in individual sediment core sections should correlate with the levels of DDT-derived compounds. Consider, for example, a core with net fine-particle accumulation rate of 0.01 cm/y. The entire history of DDT and Cs-137 accumulation would be contained in the last half centimeter of sediment that was deposited. Biological mixing to depths on the order of a few centimeters would obscure any detailed differences in the Cs-137 and DDT input histories. Mixing would be the major determinant of the depth profiles of these substances and, at sampling intervals of one to two centimeters, a very good correlation would be expected between the activity of Cs-137 and levels of DDT-derived compounds. The picture is complicated slightly by DDT geochemistry. Under anaerobic conditions, pp'-DDT can be converted to pp'-DDD (Guenzi and Beard, 1968). This can be taken into account by seeking a correlation between Cs-137 activity and the sum of pp'-DDT and pp'-DDD concentrations. Data from the upper six centimeters of core EN-123–BC-E6 and the surface sections of HSV 6, HSV7, and HSV8 are presented in Figure 3. A ratio of approximately 1 ppb (pp'-DDT + pp'-DDD) per 60 pCi/kg of Cs-137 is consistent with these data on recent deposits in Areas 2 and 3. Our surface sediment samples from near the dumpsites had much higher ratios, i.e. enhanced relative concentrations of DDT-derived compounds, which we interpret as an indicator of sources related to dredge spoil and sewage sludge disposal and Hudson-Raritan discharge. The most likely dominant source of DDT-derived compounds to Areas 2 and 3 is atmospheric deposition. Further evidence for the importance of atmospheric inputs of DDT to Area 3, the continental slope, is presented below.

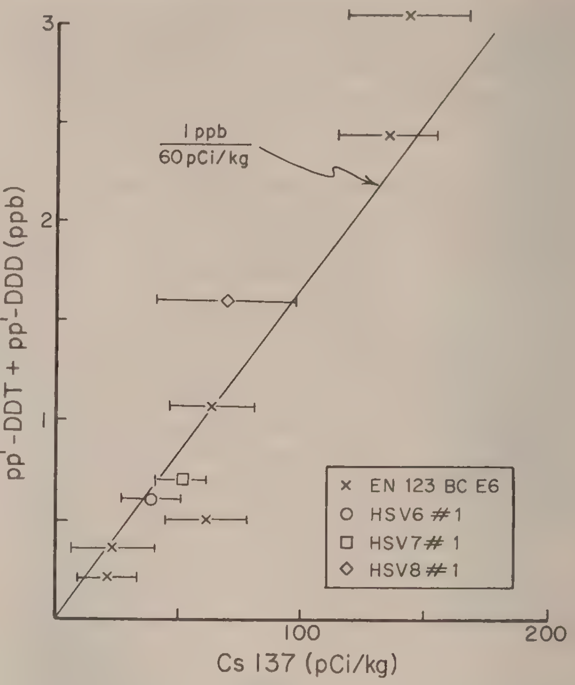

Figure 3

pp'-DDT + pp'-DDD concentrations versus Cs–137 activities in sections of sediment cores from Areas 2 and 3. The Cs–137 data are plotted with error bars of ± one standard deviation based on counting statistics.

DDT-Derived Compounds on Particles from the Continental Slope

The profile of concentrations of DDT-derived compounds in core EN-123–BC-E6 has been described previously (Anderson et al., 1988). This core, from 1640 m depth on the continental slope, is particularly valuable because of its well-defined and carefully sampled aerobic zone which provided a record of DDT deposition unaltered by anaerobic conversion to DDD. The net particle accumulation rate in the core was estimated to be on the order of 0.01 cm/y. As a result of biological mixing, DDT-derived compounds were found to a depth of about 7 cm in the core. In situ conversion of DDT to

DDD was observed at depths below 4 cm. Iron and manganese pore water chemistry indicated that the onset of anaerobic conditions occurred at about this level (Buesseler, 1986). The total historical deposition of DDT-derived compounds at this site was computed to be 48 $\mu g/m^2$ for pp'-DDT, 13 $\mu g/m^2$ for pp'-DDE, and 1.8 $\mu g/m^2$ for pp'-DDD. Such dominance of the parent compound, pp'-DDT, has been shown to be characteristic of atmospheric inputs (Rapaport et al., 1985). Concentration data from EN-123–BC-E6 indicate that such inputs can supply parts per billion levels of DDT-derived compounds to recent sediments. Other possible inputs to our study area, including Hudson-Raritan discharge and dredge spoil and sewage sludge disposal, would be vastly different and would be characterized by pp'-DDT to pp'-DDD ratios less than unity, as discussed below.

Assuming that most of the atmospheric input of DDT recorded by core EN-123–BC-E6 occurred over the approximately twenty years of major domestic use, that period would be characterized by average yearly atmospheric pp'-DDT fluxes of about 2.4 $\mu g/m^2 \cdot y$. This value falls within the range of peak atmospheric inputs estimated from dated cores from peat bogs in central and eastern North America (Rapaport et al., 1985). That study also indicated a continuing atmospheric input estimated for the early 1980's at 10–20% of peak inputs. Since the use of DDT was banned in the U.S. and Canada in 1972, the source of the early 1980's deposition was suggested to be DDT used in Mexico and Central America (Rapaport et al., 1985).

An indication of the recent delivery of DDT to the continental slope can be obtained by analyzing data from sediment trap samples collected in 1984 as part of the SEEP program (Biscaye et al., 1988). The results presented in Table 4 are from two moorings on the continental slope, one at 1250 m water depth and the other at 2300 m. The traps were deployed for 175 days, from 23 April to 15 October 1984. Typically, only a few hundred milligrams of sample were available, which made ppb-level DDT-derived compound analysis quite challenging. Measured pp'-DDT fluxes ranged from 0.07 to 0.27 $\mu g/m^2 \cdot y$ (n=7) with an average of 0.18 $\mu g/m^2 \cdot y$. This estimate of mid 1980's flux is about an order of magnitude lower than the flux during major domestic DDT use that was inferred from the data for core

Table 4
DDT and DDD in SEEP sediment trap samples.[1]

Mooring #/ Trap #	Trap depth/ Water depth (m)	pp'-DDT conc.[2] (ppb)	pp'-DDT flux[3] ($\mu g/m^2/y$)	pp'-DDD conc. (ppb)	pp'-DDD flux ($\mu g/m^2/y$)	pp'-DDT/pp'-DDD
Poisoned Traps[4]						
5/5	167/1250	2.5±0.3	0.15	1.5±0.2	0.089	1.7±0.3
5/13	872/1250	2.9±0.4	0.21	1.2±0.1	0.090	2.4±0.4
5/14	1217/1250	1.9±0.2	0.27	0.68±0.06	0.10	2.8±0.4
6/8	468/2300	2.3±0.5	0.067	1.1±0.2	0.032	2.1±0.6
6/9	868/2300	3.1±0.4	0.17	1.3±0.2	0.070	4.1±0.8
6/12	1768/2300	2.8±0.4	0.18	0.69±0.08	0.044	6.3±1.1
6/17	2263/2300	2.4±0.3	0.18	0.38±0.05	0.028	
Unpoisoned Traps[5]						
4/3	158/500	1.3±0.4		2.6±0.3		
4/20	498/500	0.1±0.1		1.3±0.1		

[1] All samples were from the summer SEEP deployment described in detail in Biscaye et al. (1988).

[2] Error estimates are based on the reproducibility of blanks and the precision of replicate analyses (about ±10%) as reported by Bopp et al. (1982).

[3] Flux calculations were based on the total mass fluxes, contaminant concentrations, trap openings (729 cm^2), and deployment times (175 days).

[4] Traps were poisoned with a 10% solution of sodium azide in filtered seawater to which sodium chloride was added. The sodium azide was used to inhibit biological activity in the traps, including the conversion of pp'-DDT to pp'-DDD. The sodium chloride was added to increase the density of the trap solution to minimize mixing with the overlying water.

[5] Because of the likelihood of conversion of pp'-DDT to pp'-DDD in the unpoisoned traps during deployment and uncertainty about the sampling period represented by the trap 20 sample, no flux calculations or computations of the pp'-DDT to pp'-DDD ratio were made for these samples.

EN-123–BC-E6 (2.4 μg/m^2·y). This decrease in pp'-DDT flux is in reasonable agreement with the peat core data cited above.

Fluxes of excess Pb-210 in the trap samples appear to vary directly with pp'-DDT flux (Fig. 4). We attribute this to similar sources and behaviors of the two substances. The excess Pb-210 originates from decay of gaseous Rn–222 in the atmosphere and, similar to DDT, Pb-210 is quite particle reactive in natural water systems. The average Pb-210 flux in these trap samples was about 1 disintegration per minute (dpm)/cm^2·y, similar to the long-term steady state average flux that would be expected for core EN-123–BC-E6 based on other cores taken along the SEEP transect (Biscaye et al., 1988). This suggests that these trap samples, on average, provide a good representation of recent accumulation at coring site EN-123–BC-E6.

The data from core EN-123–BC-E6 implied a total flux of pp'-DDT (48 μg/m^2) about 25 times greater than that of pp'-DDD (1.8 μg/m^2), whereas the trap data (Table 4) indicate the recent flux of pp'-DDT exceeds the flux of pp'-DDD by only about a factor of three. In absolute terms, the total pp'-DDD flux requires an average yearly flux on the order of 0.05 μg/m^2·y since the beginning of significant domestic DDT use in about 1950. This long-term flux estimate for pp'-DDD falls within the range of values determined for the mid 1980's flux from the sediment trap samples (0.03 to 0.10 μg/m^2·y, Table 4). This suggests that the pp'-DDD flux is an indicator of non-atmospheric input which has not experienced a major (order of magnitude) change over the past three or four decades. The cross-shelf transport of contaminants derived from river discharge or dredge spoil and sewage sludge disposal is a likely source. A relatively constant pp'-DDD flux to sediments in Area 3 would suggest that the time between delivery of pp'-DDD to Area 1 and deposition in Area 3 sediments is many years. Such long transport times or water column residence times would obscure the recent decreases in pp'-DDD levels observed in New York Harbor sediments (Bopp and Simpson, 1989). While DDT-derived compounds may provide an indicator of cross-shelf contaminant transport, such transport does not appear to be the dominant mid 1980's source of these compounds to the slope environment. The excess of pp'-DDT over pp'-DDD in the trap samples indicates that atmospheric inputs continue to be most important.

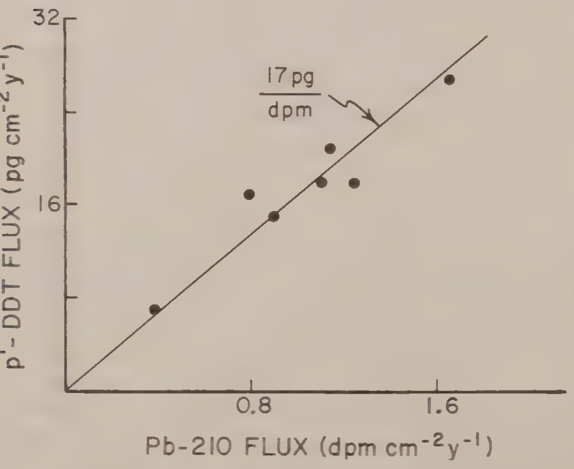

Figure 4
pp'-DDT flux versus Pb-210 flux derived from the sediment trap samples listed in Table 4.

It is possible that there was some anaerobic conversion of pp'-DDT to pp'-DDD within the traps. In an effort to guard against this, the traps were initially poisoned with a 10% sodium azide solution in filtered seawater to which sodium chloride was added in order to increase its density to minimize mixing with the overlying water. This treatment was at least somewhat successful in that it prevented the foul H_2S odor and almost complete loss of pp'-DDT that was observed in some unpoisoned traps (Table 4), however, some conversion of pp'-DDT to pp'-DDD in our samples cannot be ruled out. Even a worst case scenario, however, would not drastically alter our general interpretations of the data. If all of the pp'-DDD were produced from pp'-DDT within the traps, our estimate of recent pp'-DDT flux would increase by less than a factor of two, and the argument that the principal input of DDT-derived compounds to the slope environment is derived from the atmosphere would be strengthened. If (pp'-DDT + pp'-DDD) levels were used to determine the flux of pp'-DDT to the traps, the correlation with Pb-210 fluxes (Fig. 4) would still be very good. We hope to address the concern about conversion of pp'-DDT to pp'-DDD during trap deployment by analyzing samples of fresh suspended particles from the shelf and slope that would be collected by in situ filtration.

Sources of DDT-Derived Compounds to Nearshore Particles

Data related to the pp'-DDT to pp'-DDD ratio in Hudson-Raritan discharge, sediments dredged from the N. Y. Harbor complex, and New York City sewage sludge are presented in Table 5. We interpret the suspended matter samples as representative of Hudson-Raritan discharge. New York City sewage sludge is represented by a single sample from the Ward's Island treatment plant collected in the mid 1970's. Data on samples of raw sewage and primary treated effluent from the Yonkers, N.Y., sewage treatment plant are also included. These

Table 5

Ratio of pp'-DDT to pp'-DDD in selected samples.

Sample	pp'-DDT / pp'-DDD
Suspended Particles[1]	
N.Y. Harbor, Hudson R. (GFF1060, 1981)	0.22
Newark Bay (F1049, 1985)	0.22
Newark Bay (F1054, 1985)	0.11
Newark Bay (F1077, 1986)	0.22
Newark Bay (F1091, 1986)	0.16
Arthur Kill (GFF1057, 1981)	0.19
Arthur Kill (F1089, 1986)	0.44
Kill van Kull (F1090, 1986)	0.25
Raritan Bay (GFF1055, 1981)	0.63
Sediment Cores from the N.Y. Harbor Complex[2]	
–1.7 (CN1472, N.Y. Harbor, Hudson R.)	0.05
NB13 (CN1964, Newark Bay)	0.18
RB17 (CN1533, Raritan Bay)	0.16
KvK1 (CN1959, Kill van Kull)	0.87
Sewage Samples	
Secondary sludge from the NYC treatment plant on Ward's Island	0.2
Raw sewage from the Yonkers, N.Y.[3] sewage treatment plant	0.3
Primary treated effluent from the Yonkers, N.Y. sewage treatment plant	0.2

[1] Suspended particle samples are identified by the Lamont control numbers given in parentheses.

[2] The ratio reported is the average pp'-DDT level in the core sections divided by the average pp'-DDD level. Sediment cores are identified by Lamont control numbers given in parentheses. Control Numbers of cores collected for this study are given in Appendix Table 1.

[3] The city of Yonkers is located on the east bank of the Hudson and is bordered on the south by NYC.

data can be used to investigate the possible sources of DDT-derived compounds to the sediment cores from Area 1 and to our suspended particle samples.

From the data in Table 3, we estimate that the surface sediment samples from our Area 1 cores had an average pp'-DDT to pp'-DDD ratio of about 0.5. It is tempting to cite the much lower average pp'-DDT to pp'-DDD ratio in cores 1.7, NB 13, and RB17 (Table 5) as evidence that atmospheric inputs contribute significantly to the DDT-derived compounds in Area 1 sediments and that dredge spoil disposal was not the dominant source of contaminants to our Area 1 cores. We believe, however, that this argument is seriously flawed. Sediment samples from the N.Y. Harbor complex waters bordering Staten Island had the highest levels of pp'-DDT that we have observed in N.Y. Harbor sediments and a relatively high ratio of pp'-DDT to pp'-DDD (core KvK1, Table 5). In the case of the Kill van Kull, which forms the northern border of Staten Island, sections of a core that records deposition from approximately the mid 1970's to 1985 had an average pp'-DDD level of 580 ppb and an average pp'-DDT of 500 ppb (Bopp and Simpson[1]). Comparable results have been obtained from sediment core samples from the Arthur Kill, on the western boundary of Staten Island (Bopp, unpublished data). To put this in perspective, sediment deposited along the main axis of the Hudson near the site of core 1.7 from approximately the mid 1970's to 1986 had an average pp'-DDD level of approximately 50 ppb (core 1.65, CN1923, Bopp and Simpson, 1989) and an average pp'-DDT level of approximately 5 ppb (Bopp, unpublished data). We have found no published information that indicates the source of the relatively high levels of DDT-derived compounds in recent Kill van Kull and Arthur Kill sediments. From the above discussion, it is evident that even a small component of sediment similar to that found in the Kill van Kull core would significantly increase the ratio of pp'-DDT to pp'-DDD in material dredged from the N. Y. Harbor complex. Data from the U.S. Army Corps of Engineers indicates that in 1986, projects in the Arthur Kill and Kill van Kull contributed about 14% of the volume of material disposed of at the dredge spoil dumpsite.[3] This indicates that the pp'-DDT to pp'-DDD ratio in the surface sediment samples from our Area 1 cores does not require a significant atmospheric input nor does it rule out the possibility that dredged material disposal is the dominant source of these compounds to the sediments.

Strong evidence for atmospheric inputs of DDT-derived compounds to Area 1 was found in the pp'-DDT to pp'-DDD ratios in our suspended particle samples taken in Area 1 (Table 6). The fact that major inputs of DDT-derived compounds to Area 1 with dredge spoil and sewage sludge disposal and Hudson-Raritan discharge did not totally obscure the atmospheric signal underscores the power of the pp'-DDT to pp'-DDD ratio as a tracer in the coastal environment. Figure 5 shows station locations and all of the pp'-DDT to pp'-DDD ratio data. The majority of the data is easily interpreted. At the G1 station, near Sandy Hook buoy G1, the ratio is consistent with Hudson-Raritan discharge. It is likely that this source continues to be the major influence at site SB (at the bell buoy off Shrewsbury Rocks) which is located within the plume of discharge from the Hudson-Raritan that follows the New Jersey coast (Ingham, 1989). Further south along the coast, as

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[3] U.S. Army Corps of Engineers, New York District, Mud Dump Activity Listout, 1 December 1990.

Table 6
Large volume filter samples.

Station[1]	Suspended Matter (mg/l)	pp’-DDT[2] (ppb)	pp’-DDD (ppb)	pp’-DDT/pp’-DDD
August 24, 1987				
G1 (S), F1099	2.7	4.8±0.7	25±3	0.19±0.03
A54 (S), F1097	0.16	24±3	7.2±1.5	3.3±0.8
A54 (D), F1098	1.4	5.2±0.6	14±2	0.36±0.06
LB3 (S), F1094	0.41	14±2	10±1	1.4±0.3
LB5 (S), F1095	0.19	15±2	11±2	1.4±0.3
A56 (S), F1096	0.15	17±3	6.3±2.0	2.7±1.0
October 31, 1987				
G1 (S), F1103	1.5	5.8±0.7	18±2	0.33±0.05
A44 (D), F1102	0.80	6.9±1.0	9.9±1.1	0.70±0.13
SB (D), F1100	1.3	8.7±1.1	22±2	0.40±0.07
A54 (D), F1101	1.0	[3]	[3]	0.23
June 27, 1988				
G1 (S), F1105	1.3	16±2	36±4	0.43±0.07
G1 (D), F1106	1.4	9.6±1.1	25±3	0.38±0.06
SB (S), F1107	0.87	9.3±1.4	31±3	0.30±0.06

[1] Station locations are shown on Fig. 5. A44, A54 and A56 are NOAA Broadscale designations. LB3 and 5 are along the NOAA Long Branch transect. G1 is green buoy #1 off Sandy Hook and SB is the bell buoy off Shrewsbury Rocks. (S) indicates surface samples collected at a depth of about 2 meters; (D) are deep samples collected at about 5 meters above bottom. F designations are Lamont control numbers used for sample identification.

[2] Error estimates are based on the reproducibility of blanks and the precision of replicate analyses (about ±10%) as reported by Bopp et al. (1982).

[3] Individual concentrations are not reported due to an uncertainty in the total extraction volume.

the plume is diluted by mixing, there is evidence of an additional source of DDT-derived compounds. At sites LB3 and LB5 (Long Branch) the pp’-DDT to pp’-DDD ratio increases to greater than one as a result of higher levels of pp’-DDT. Near the dumpsites at stations A44 and A54, the pp’-DDT to pp’-DDD ratios in the deep suspended particle samples are similar to those in near-surface sediments from our Area 1 cores (Table 3). We interpret this as an indication of resuspension. The pp’-DDT to pp’-DDD ratios would also be consistent with a Hudson-Raritan discharge source, although the great difference between deep and surface samples makes this unlikely. Surface suspended particle samples at sites A54 and A56 had the highest levels of pp’-DDT and ratios of pp’-DDT to pp’-DDD much greater than one (Table 6). It should be noted that all of the samples used to characterize Hudson-Raritan discharge, dredge spoil disposal, and sewage sludge disposal had pp’-DDT to pp’-DDD ratios of less than one (Table 5) and so an additional source is indicated.

As mentioned above, a most likely source of enhanced pp’-DDT is atmospheric input. The ratios of pp’-DDT to pp’-DDD in the surface suspended particle samples from sites A54 and A56 were similar to those observed in sediment trap samples from the continental slope (Table 4). However, the concentration of pp’-DDT (Table 6) on these suspended particle samples was much higher than in the trap material. The surface suspended particles at A56 and A54 had about 20 ppb pp’-DDT while the trap samples averaged about 2.5 ppb. This could reflect differences in sample type (traps collect the larger, sinking particles), in location and collection period, or perhaps in pp’-DDT sources. Analysis of suspended particle samples from the continental slope environment would help to limit these possibilities.

For Area 1, our data indicate that the contaminant regime in surface waters can be vastly different than that found near the bottom. The pp’-DDT to pp’-DDD ratio on particles appears to have significant potential in the nearshore environment as a tracer of water masses and the fate of particle-associated contaminants. Specific application to onshore transport of pollutants and deposition in coastal embayments and estuaries as discussed by Olsen et al. (1989) is indicated. At this point, however, it must be remembered that our interpretations are based on a limited number of samples and that other possible sources of pp’-DDT have not been ruled out. While the collection and analysis of addi-

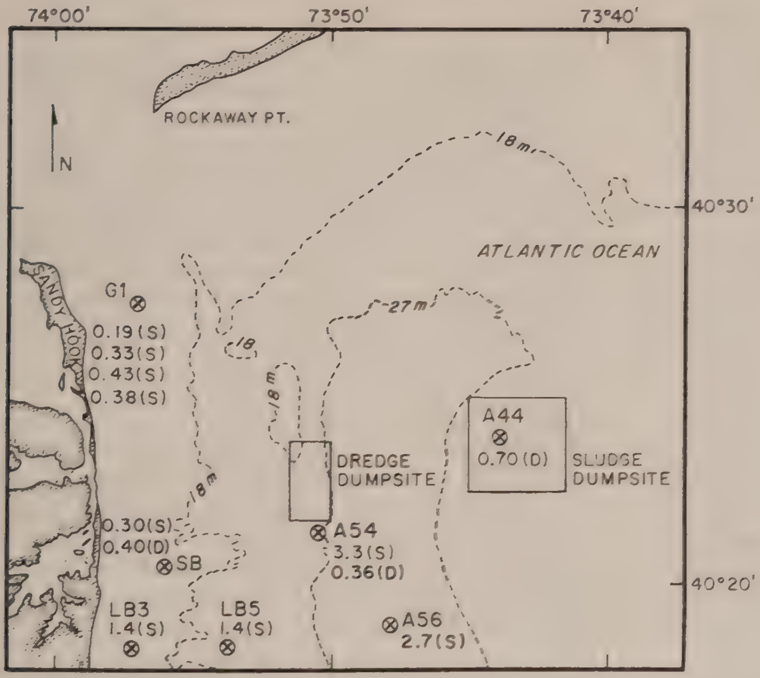

Figure 5
pp'-DDT to pp'-DDD ratio on suspended particle samples from Area 1. S = indicates surface samples collected approximately 2 m below sea surface. D = indicates deep samples collected at approximately 5 m above bottom.

Table 7
2,3,7,8-tetrachlorodibenzo-p-dioxin concentrations in area 1 samples.

Sample	2,3,7,8-TCDD (parts per trillion)
Core Sections	
A41 #1, 0–20 cm	400
A56 #4, 0–2 cm	41
A56 P1, 72–80 cm	ND(18)[1]
Suspended Matter	
G1 (S), F1099	120
A54 (D), F1098	ND(33)

[1] ND indicates not detected and is followed by the detection limit in parentheses.

tional samples is necessary, the potential for increasing our understanding of pollutant behavior in this important environment indicates that such an effort is warranted.

2,3,7,8-TCDD on Particles from Area 1

A major incident of 2,3,7,8-Tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD) contamination in sediments of Newark Bay has been described recently (Bopp et al., 1991). It was estimated that sediments deposited in the bay over the past four decades averaged between 600 and 1200 parts per trillion (ppt) 2,3,7,8-TCDD. This study indicated that discharges from an industrial chemical manufacturing plant on the lower Passaic River (a major tributary of Newark Bay) could account for much of the 2,3,7,8-TCDD contamination in the N. Y. Harbor complex. Major inputs of 2,3,7,8-TCDD to Area 1 associated with Hudson-Raritan discharge and Newark Bay dredge spoil disposal would be expected. The data in Table 7 confirms the presence of significant levels of 2,3,7,8-TCDD on particles from Area 1.

Based on our analyses, it appears possible that levels of 2,3,7,8-TCDD could provide a specific indicator of the dredged material disposal source to the sediments of Area 1. The 120 ppt level in the 1987 sample of suspended matter from near Sandy Hook at the mouth of the Hudson-Raritan estuary is an indicator of mid 1980's input from discharge. Samples of mid 1980's particles from the vicinity of the dumpsites included the deep suspended matter sample from site A56 that contained <33 ppt 2,3,7,8-TCDD and the 0–2 cm section of core A56 #4 that contained 40 ppt. This sediment core section was among those mentioned above that contained detectable levels of Be-7 indicating deposition within about a year prior to collection (1987). The concentration of 400 ppt 2,3,7,8-TCDD found in the composite sample from site A41 indicates that past inputs were significantly greater, an observation supported by the Newark Bay sediment data and chemical production records of the industrial facility (Bopp et al., 1991). Most interesting was the low level of 2,3,7,8-TCDD in the 72–80 cm sample from piston core P1 at site A56. In this particular core, Cs-137 penetrated to 88 cm (Table 2). The 72–80 cm sample contained a significant recent component (115±16 pCi/kg of Cs-137). This core section represents an attempt to sample sediment from approximately the late 1950's. Bopp et al. (1991) indicate that, compared to the mid 1980's, the period between the mid 1950's and early 1960's would be characterized by much higher levels of 2,3,7,8-TCDD contamination, both in sediments depositing in Newark Bay and the lower Passaic River and in Hudson-Raritan discharge derived from the bay. It would be difficult to explain the low level of 2,3,7,8-TCDD in the sample under consideration if Hudson-Raritan discharge were the major contributor to the contamination observed in our cores from Area 1. Our observations are consistent with dredged material disposal being a major source. Prior to 1970, material dredged from Newark Bay was disposed of primarily in upland sites adjacent to the bay (Suszkowski, 1978). Only since 1970 has the dumpsite located in our study area been the dominant repository for Newark Bay dredged materials (Suszkowski, 1978; HydroQual, Inc.[4]). If A56 P1 (72–80 cm) does indeed represent dredged material dating from approximately the late 1950's, it would not have a large Newark Bay component and would be expected to contain a relatively low level of 2,3,7,8-TCDD, consistent with value reported (Table 7). At present, this interpretation is quite speculative due to the limited amount of applicable data.

Conclusions

The conclusions of this study can be summarized as follows:

1) High rates of fine particle and associated pollutant accumulation were found along the axis of the HSV from the vicinity of the dredge spoil and former sewage sludge dumpsites to sites several kilometers down-valley. The high rates of net particle accumulation, data on 2,3,7,8-TCDD, and trends in Cs-137 and K-40 activities in sediments are consistent with the conclusion that the major source of accumulating particles is the translocation of material dredged from the N.Y. Harbor complex and dumped at the dredge spoil disposal site. Other studies (Davis et al., 1995) indicate that sewage sludge derived material has also been deposited at these sites. Any estimate of total recent particle and contaminant deposition down valley from the dumpsites would require additional sediment cores from transects across the axis of the HSV to complement our along-axis samples.

2) Cs-137 and K-40 activities and contaminant analyses in sediment core sections indicate that fine particles that have accumulated in a mid-shelf region (Area 2) and on the continental slope (Area 3) over the past four decades are relatively uncontaminated. Analyses of DDT-derived compounds in these samples indicate a dominantly atmospheric source. Cumulative and mid 1980's fluxes of DDT-derived compounds calculated from sediment trap and core data are consistent with other estimates of regional atmospheric inputs (Rapaport et al., 1985).

3) The pp'-DDT to pp'-DDD ratio on particles is a useful tracer of atmospheric inputs and processes associated with particle-reactive pollutant behavior in the continental slope and coastal environments. Indications of atmospheric inputs have been found on suspended particles collected within a few kilometers of the Hudson-Raritan discharge and the dredge material and former sewage sludge disposal sites. Further exploitation of this tracer is recommended.

4) Based on four samples, there appears to be significant 2,3,7,8-TCDD contamination on particles from Area 1 that is probably related to industrial discharges to the lower Passaic River. Additional samples should be analyzed to test this provisional conclusion.

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Appendix Table 1

Radionuclide data on sediment core sections.[1]

Location	Depth (cm)	Weight (g)	Cs-137 (pCi/kg)	Be-7 (pCi/kg)	K-40 (pCi/g)
NY5 #1 CN 2032 10/30/87	0–2	51.2	49±14	167±103	10.9±0.9
	2–4	51.1	75±14	–26±104	10.8±0.6
	4–6	62.4	66±16		10.7±0.6
	6–8	67.9	56±15		10.2±0.6
	8–12	129.6	73±8		10.7±0.6
	12–16	133.2	21±7		10.9±0.6
	16–20	130.3	11±6		11.3±0.6
	20–24	130.9	0±6		10.7±0.6
NY5 #2 CN 2033 10/30/87	0–0.5	15.0	50±20	178±209	11.6±0.7
NY 6 #1 CN 2030 10/30/87	0–2	40.0	54±12	195±95	8.6±0.5
	2–4	67.5	37±7	7±60	8.3±0.4
	4–6	70.0	47±10		8.1±0.4
	6–8	66.7	37±14		8.7±0.5
	8–12	145.1	29±9		8.7±0.5
	12–14	66.9	8±7		9.0±0.5
NY6 #2 CN 2031	0–0.5	7.4	55±31	335±341	10.2±0.7
	0.5–1.0	8.5	75±30	–33±338	10.7±0.7
A25 #1 CN 2034 10/30/87	0–2	26.3	418±31	127±143	16.7±0.9
	2–4	31.3	536±35	276±149	17.8±1.0
	4–6	31.0	230±22	21±123	15.3±0.8
	6–8	26.1	370±33		15.7±0.9
	6–8[2]	26.1	403±32		17.5±1.0
	6–8[2]	26.1	362±25		18.1±1.0
	6–8[2]	26.1	318±30		17.8±1.0
	8–12	51.2	419±29		17.6±1.0
	8–12[2]	51.2	420±29		17.2±0.9
	12–16	56.6	320±25		17.3±0.9
	16–20	55.9	163±18		17.4±0.9
	16–20[2]	55.9	192±20		17.6±1.0
	20–24	61.4	198±19		18.2±1.0
	24–28	57.9	234±23		18.9±1.0
	28–32	66.4	126±12		15.6±0.8
	28–32[2]	66.4	127±15		15.5±0.8
	32–33	29.1	220±18		13.2±0.7
A25 #2 CN 2035 10/30/87	0–0.5	11.8	201±31	33±261	14.9±0.9
A25 P[3] CN 2075 10/4/89	0–4	77.6	27±7	–11±61	10.6±0.6
	4–8	57.8	23±9		9.0±0.5
	8–12	55.0	45±10		9.5±0.5
	12–16	55.8	21±10		10.1±0.6
	16–24	60.6	192±14		14.1±0.7
	24–32	58.2	180±15		12.5±0.7
	32–40	90.4	39±7		7.6±0.4
	40–48	60.9	321±19		12.7±0.7

Continued

Appendix Table 1 (continued)

Location	Depth (cm)	Weight (g)	Cs-137 (pCi/kg)	Be-7 (pCi/kg)	K-40 (pCi/g)
A41 #1	0–2	22.8	129±18	70±123	12.5±0.7
CN 2028	2–4	28.6	227±21	–77±120	12.9±0.7
10/30/87	4–6	19.0	387±37	17±369	14.5±0.9
	6–8	17.3	394±34		16.1±0.9
	6–8[2]	17.3	420±46		15.4±1.0
	6–8[2]	17.3	351±34		15.7±0.9
	8–12	39.3	342±28		13.2±0.8
	12–16	40.8	361±30		14.0±0.8
	16–20	51.0	180±18		14.6±0.8
	20–24	80.4	13±11		12.8±0.7
	24–28	58.4	10±13		15.8±0.9
	28–32	51.5	–10±11		16.9±0.9
	32–36	60.8	–5±9		15.8±0.8
	36–40	99.4	17±11		11.7±0.6
	40–44	118.0	–7±9		11.5±0.6
	44–47	111.8	4±7		11.2±0.6
A41 #2 CN 2029 10/30/87	0–0.5	4.4	162±60	–369±759	13.9±1.1
A56 #1	0–2	16.3	151±25	386±205	13.0±0.8
CN 2024	2–4	18.3	214±34	109±240	13.7±0.9
9/1/87	4–6	20.1	256±36	–55±236	14.3±0.9
	6–8	21.6	229±29		12.9±0.8
	8–12	36.8	158±18		12.0±0.7
	40–42	18.2	196±26		14.6±0.9
A56 #2	0–2	16.2	157±20	1008±179	14.3±0.8
CN 2025	2–4	13.9	191±29	65±244	15.6±0.9
9/1/87	4–6	14.6	173±20	–12±211	14.1±0.8
	6–8	15.4	206±28		13.9±0.8
	40–42	17.5	228±35		12.9±0.9
A56 #3	0–2	15.3	180±38	610±239	13.8±0.8
CN 2026	2–4	18.8	334±34	–200±252	14.0±0.9
9/1/87	4–6	20.0	188±30	–11±253	13.3±0.9
	6–8	18.8	217±29		13.1±0.8
	8–12	32.7	304±25		16.2±0.9
	12–16	32.2	278±25		13.5±0.8
	16–20	40.1	280±19		12.8±0.7
	20–24	32.7	267±25		11.8±0.7
	24–28	30.4	408±30		12.0±0.7
	28–32	29.9	338±28		14.6±0.8
	32–36	36.8	252±23		14.2±0.8
	36–40	53.0	148±13		13.1±0.7
	40–44	42.0	191±22		12.4±0.7
	44–47	36.4	77±15		13.2±0.7
A56 #4	0–2	15.2	186±21	636±193	14.2±0.8
CN 2027	2–4	15.8	249±29	–182±234	14.2±0.9
9/1/87	4–6	24.3	194±20		13.4±0.8
	6–8	27.3	139±17		13.5±0.7
	8–12	36.1	219±19		13.7±0.8
	12–16	34.8	225±23		12.6±0.8
	16–20	34.8	285±25		13.8±0.8

Continued

Appendix Table 1 (continued)

Location	Depth (cm)	Weight (g)	Cs-137 (pCi/kg)	Be-7 (pCi/kg)	K-40 (pCi/g)
A56 #4	20–24	30.1	365±30		13.4±0.8
CN 2027	24–28	32.4	348±26		13.4±0.7
9/1/87	28–32	34.2	260±20		14.8±0.8
	32–36	41.3	225±19		13.9±0.7
	36–40	35.4	209±17		13.8±0.7
	40–44	39.7	168±19		11.8±0.7
	44–48	44.5	123±17		12.8±0.7
A56 #5	0–2	17.9	174±23	302±191	15.4±0.9
CN 2053	2–4	16.7	173±21	–216±177	15.0±0.8
9/26/88	4–6	20.1	229±26		15.2±0.9
	6–8	23.1	203±20		14.2±0.8
	12–16	34.4	213±23		13.7±0.8
	16–20	31.6	279±26		15.2±0.9
	20–24	31.2	282±27		14.6±0.9
	24–28	37.5	233±23		14.3±0.8
	28–32	39.1	272±25		15.0±0.9
	32–36	34.4	274±26		13.5±0.8
	36–40	29.1	326±30		13.4±0.8
	40–43	27.6	371±30		13.4±0.8
A56 #6	0–2	14.8	137±24	702±174	14.9±0.9
CN 2076	2–4	17.5	133±23	268±206	12.1±0.7
10/4/89					
A56 P1[4]	0–4	20.7	279±29	–274±210	14.6±0.8
CN 2052	4–8	8.8	327±37		14.0±0.9
9/26/88	8–12	9.8	312±38		12.9±0.8
	12–16	9.5	422±42		14.6±0.9
	16–24	22.4	429±29		14.1±0.8
	24–32	20.6	300±27		16.4±0.9
	32–40	20.6	261±26		15.4±0.9
	40–48	25.3	207±20		15.6±0.8
	48–56	19.9	194±24		15.5±0.9
	56–64	25.3	184±21		13.5±0.8
	64–72	26.2	157±20		13.8±0.8
	72–80	31.6	115±16		15.0±0.8
	80–88	28.2	36±18		17.0±0.9
	88–96	34.4	28±16		16.8±0.9
	96–104	35.9	–13±13		16.5±0.9
	104–112	34.3	18±16		16.7±0.9
	112–120	38.1	–16±13		15.7±0.9
	136–144	53.9	–4±10		14.0±0.7
	160–168	46.7	7±11		14.6±0.8
A56 P2	0–4	10.1	158±47		17.0±1.2
CN 2077	4–8	12.8	188±43		13.3±1.0
10/4/89	8–12	19.8	110±28		13.2±0.8
	12–16	14.3	101±33		12.8±0.9
	16–24	22.6	215±29		13.0±0.8
	24–32	18.2	251±34		14.6±0.9
	32–40	19.8	229±29		15.5±0.9
	40–48	21.4	235±30		14.9±0.9
	48–56	23.5	290±30		14.8±0.9
	56–64	17.5	262±32		14.5±0.9
	64–72	23.3	304±23		14.6±0.8

Continued

Appendix Table 1 (continued)

Location	Depth (cm)	Weight (g)	Cs-137 (pCi/kg)	Be-7 (pCi/kg)	K-40 (pCi/g)
A56 P2	72–80	22.1	311±26		14.7±0.8
CN 2077	80–88	16.1	315±28		13.7±0.8
10/4/89	88–96	14.5	315±33		13.7±0.8
(continued)	96–104	18.2	314±28		13.2±0.8
	104–112	24.6	415±28		13.4±0.7
	112–120	25.7	386±28		15.1±0.8
	120–128	21.5	262±22		14.6±0.8
	128–136	20.6	206±23		14.0±0.8
	136–144	22.2	201±22		15.7±0.9
	144–152	22.0	170±22		15.3±0.9
	152–160	21.5	192±23		14.5±0.8
	160–168	28.8	162±18		13.1±0.7
	168–176	40.2	63±11		11.6±0.6
	176–184	26.7	182±21		13.8±0.8
	184–192	25.2	92±17		14.2±0.8
	192–200	31.0	143±17		14.2±0.8
	200–208	26.1	24±16		15.3±0.8
	208–216	33.2	22±12		15.6±0.8
	216–224	38.3	–6±13		15.6±0.8
	224–232	40.1	10±11		14.2±0.8
	232–240	31.3	–2±13		16.2±0.9
	240–248	30.7	–5±14		16.8±0.9
	248–256	52.0	3±11		14.0±0.8
HSV1 #1	0–2	24.5	202±18	180±140	14.7±0.8
CN 2042	2–4	19.7	394±31	–311±279	16.9±1.0
6/27/88	4–6	25.0	423±29		17.2±0.9
	6–8	24.8	382±25		17.0±0.9
	8–12	43.4	435±32		15.7±0.9
	12–16	57.7	314±23		15.0±0.8
	16–19	42.6	228±22		15.2±0.9
HSV1 #2	0–2	25.9	339±24	172±167	17.1±0.9
CN 2043	2–4	20.5	444±38	258±380	16.4±1.0
6/27/88	4–6	22.5	447±35		17.3±1.0
	6–8	21.5	453±35		16.0±0.9
	8–12	47.1	409±24		16.2±0.8
	12–16	48.7	360±28		15.7±0.9
HSV2 #1	0–2	26.9	130±16	86±126	13.8±0.7
CN 2040	2–4	25.4	156±19	–181±217	15.0±0.8
6/27/88					
HSV2 #2	0–2	23.0	132±17	159±159	15.1±0.8
CN 2041	2–4	25.5	160±25	133±311	15.1±0.9
6/27/88	4–6	25.9	189±20		14.9±0.8
	6–8	25.4	151±20		14.9±0.8
	8–12	62.1	202±18		13.7±0.8
	12–16	71.9	141±15		12.6±0.7
	16–18	39.6	56±11		12.2±0.7
	18–21	93.0	121±13		12.3±0.7
HSV2 #3	0–2	32.4	70±13		12.3±0.7
CN 2078	0–2[2]	32.4	60±14	61±109	12.4±0.7
10/4/89	2–4	37.9	67±13	–101±123	12.5±0.7
	2–4[2]	37.9	66±16		14.2±0.8

Continued

Appendix Table 1 (continued)

Location	Depth (cm)	Weight (g)	Cs-137 (pCi/kg)	Be-7 (pCi/kg)	K-40 (pCi/g)
HSV2 #3	4–6	40.6	106±17		13.7±0.8
CN 2078	6–8	35.6	138±15		12.8±0.7
10/4/89	8–12	70.6	155±14		13.0±0.7
(continued)	12–16	90.9	90±9		11.4±0.6
	16–20	108.1	58±8		10.9±0.6
	20–24	122.5	28±8		11.7±0.6
	24–28	124.0	16±6		10.4±0.5
	28–32	123.1	15±7		11.0±0.6
	32–36	126.8	4±7		11.6±0.6
HSV3 #1	0–2	23.7	105±16	417±132	14.7±0.8
CN 2038	2–4	24.3	115±17	166±191	14.6±0.8
6/27/88	4–6	29.9	145±21		13.7±0.8
	6–8	30.7	155±21		13.7±0.8
	8–12	66.3	116±18		11.5±0.7
	12–16	73.2	104±16		12.0±0.7
	16–19	67.6	98±15		12.2±0.7
	19–21	47.7	106±16		11.7±0.7
HSV3 #2	0–2	25.7	134±16	272±142	15.0±0.8
CN 2039	2–4	27.0	147±19	−2±214	12.6±0.7
6/27/88	4–6	28.7	134±16		14.7±0.8
	24–26	53.9	53±9		11.7±0.6
HSV4 #1	0–2	48.0	53±10	118±78	11.3±0.6
CN 2059	2–4	56.2	57±9	−35±77	11.1±0.6
11/19/88					
HSV4 #2	0–2	39.6	48±11	156±100	12.4±0.7
CN 2060	2–4	41.1	77±11	54±98	12.3±0.7
11/19/88	4–6	52.4	50±10		12.2±0.6
	6–8	55.0	72±11		12.4±0.7
	8–12	108.6	56±10		10.5±0.6
	12–16	117.0	40±10		10.2±0.6
	16–20	119.2	36±11		10.4±0.6
	20–24	122.4	−9±9		10.0±0.5
HSV5 #1	0–2	43.9	53±13	−39±103	11.7±0.6
CN 2061	2–4	32.2	107±18	15±146	12.5±0.7
11/19/88	4–6	41.1	77±12		11.8±0.6
	6–8	54.5	68±10		11.2±0.6
	8–12	104.6	54±10		10.4±0.6
	12–16	113.8	36±7		10.4±0.5
	16–20	127.3	27±9		9.8±0.5
	20–24	131.2	20±7		9.5±0.5
	24–28	143.6	3±6		9.4±0.5
	28–28.5	31.1	15±13		11.9±0.7
HSV6 #1	0–2	50.6	40±12	5±97	10.8±0.6
CN 2062	2–4	59.2	50±10	−16±87	11.3±0.6
11/19/88	4–6	72.3	23±8		10.8±0.6
	6–8	71.1	12±8		10.7±0.6
	8–12	145.7	6±6		9.8±0.5
	12–16	134.6	−4±6		10.5±0.5

Continued

Appendix Table 1 (continued)

Location	Depth (cm)	Weight (g)	Cs-137 (pCi/kg)	Be-7 (pCi/kg)	K-40 (pCi/g)
HSV6 #2	0–2	56.2	35±7	–32±73	12.0±0.6
CN 2063	2–4	60.0	20±9		11.6±0.6
11/19/88	4–6	74.0	22±7		11.1±0.6
	6–8	77.1	7±7		11.1±0.6
	8–12	149.8	–5±6		10.5±0.5
HSV7 #1	0–2	54.5	52±10	54±102	11.7±0.6
CN 2064	2–4	50.9	42±13		11.5±0.6
11/19/88	4–6	59.2	42±12		11.4±0.6
	6–8	66.0	42±11		11.2±0.6
	8–12	125.6	34±9		11.0±0.6
	12–16	142.7	16±6		10.9±0.6
	16–20	139.5	15±7		11.2±0.6
HSV8 #1	0–1	10.0	70±28	91±284	18.2±1.1
CN 2065	1–2	12.3	63±25	106±259	17.5±1.0
11/19/88	2–3	12.2	88±26	148±306	19.3±1.1
	3–4	11.8	124±28		20.1±1.1
	4–6	22.4	80±17		18.5±1.0
	6–8	24.9	–15±24		17.1±1.0
HSV9 #1	0–1	8.7	4±49	358±455	17.0±1.2
CN 2066	1–2	10.7	33±41	–298±396	18.4±1.2
11/19/88	2–3	9.8	0±31		20.4±1.2
NBS Std.[5]			2675±136	14.7±0.8	
			2643±133	14.3±0.7	
			2656±134	14.5±0.7	

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[1] Activities are reported with error bars of ± one standard deviation based on statistical counting errors.

[2] These are results of replicate analyses.

[3] Site A25 is on the dredge spoil disposal site. The relatively low Cs-137 and K-40 activities in the top 40 cm of A25 P probably result from emplacement of a sand cap at that site after our collection of cores A25 #1 and 2 (10/30/87) and prior to collection of the piston core (A25 P, 10/4/89). Capping with relatively clean sand is known to have been performed at times at this disposal site. This procedure is an attempt to stabilize contaminated dredge spoil deposits and isolate them from the overlying water.

[4] This piston core overpenetrated. Based on the other cores from this site, we estimate that the top 60–120 cm were lost.

[5] These results are for NBS standard reference material 4350 (river sediment). NBS reported activities are 2703±122 pCi/kg for Cs-137 and 14.6±1.3 pCi/g for K-40.

Audience Questions

Question: I wonder if you have any idea where the atmospheric DDT comes from?

R. Bopp: Yes. It doesn't look like recycled DDT. If it were recycled from soil applications, it would have significant DDD associated with it. What Rappaport et al. (1985) suggested was that the current source was from spraying in Mexico and Central America and atmospheric transport as nearly pure DDT. Our study looked at areas near major sewage sludge and dredge spoil dumpsites and we saw that influence in near-bottom suspended particle samples and in the sediments. It's astounding that the DDT signal in the surface water right above the dumpsites at some times of the year appears to be dominated by an atmospheric signal. I find that incredible.

Question: Did you say that you had data from deep ocean sites?

R. Bopp: At the shelf break. When you get to the deep ocean, there is little net deposition. You only have to look at the top few centimeters, and you can find cores where this layer is aerobic. The DDT that gets there doesn't get converted on site. And, yes, we have a superb core from about 1,700 meters depth. So, we don't have deep ocean data, but from the shelf break at 1700 meters.

Question: With respect to the DDT compounds found in that core, how long ago would you say they were laid on the surface of the ocean?

R. Bopp: The entire history of DDT deposition is contained within the top four centimeters. The first possible input would be 1939, but the maximum inputs would have been in the fifties and sixties.

The only other thing that I'd like to leave you with is the power of the atmospheric tracer, the DDD to DDT ratio. You could be sitting on the top of the dredge spoil sites in August 1987 looking at the particles that are in that water column and barely recognize the influence of either the disposal sites or the Hudson/Raritan discharge in terms of the DDT signature. I was amazed to find that it's predominantly atmospheric.

Introduction

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The coastal waters in the apex of the New York Bight (NYB) receive enormous anthropogenic inputs generated by the domestic and industrial activities of approximately 20 million people. During the early 1980's, before the closure of the 12-mile sewage sludge dumpsite (Fig. 1), the three largest sources of pollution to the apex were the outflow from the Hudson–Raritan estuary, and the dumping of sewage sludge and dredged material from barges. Relatively minor inputs came into the apex from coastal New Jersey and Long Island and from aerial or atmospheric deposition (Fig. 2).

For some wastes, such as nitrogen and organic carbon, the Hudson–Raritan estuarine plume overwhelmed the contribution from other sources during the early 1980's. For example, approximately 58% of the anthropogenic organic carbon was from the estuarine plume, and 5% from sewage sludge (Mueller et al., 1976; Fig. 2). For other inputs such as lead, dredged material was by far the largest source at 51%, while sewage sludge provided 21% and the estuary about 24% of the loadings (Stanford and Young, 1988). The contribution of sewage sludge to total loadings of PCB's was relatively minor in comparison with inputs from the estuary and dredged material (Fig. 2). Inputs of colliform bacteria from the plume were more than 100 times the inputs released by dumping of sewage sludge (Mueller et al., 1976). Inputs of the major contaminants from sewage sludge were generally subordinate to those from the estuary and from dumping of dredged material.

While the major sources of pollution to the apex were reasonably well quantified, their fates in the NY Bight were not well understood. A number of earlier studies (e.g. Hatcher and McGillivary, 1979; Zdanowicz, 1991) documented highly elevated concentrations of sewage-derived contaminants in the sediments of the Christiaensen Basin, a topographic depression (the Hudson Shelf Valley) between the sewage sludge and dredged material dumpsites (Fig.1). The proximity of the dumpsites and the mouth of the Hudson–Raritan Estuary to the basin makes these sources potentially significant. Further, resolution of their relative importance to the habitat quality of the Christiaensen Basin becomes even more difficult because sewage sludge, dredged materials, and the estuarine plume share many of the same sewage-associated contaminants. Young et al. (1985) suggest that the highly dispersive circulation in the NYB results in particle-associated contaminants from a variety of sources seeking the same naturally muddy sediments, such as those in the Christiaensen Basin, regardless of their point of introduction.

Studies of Landsat satellite images of turbidity (Fedosh and Munday, 1982) indicate that the seaward edge of the estuarine plume extends over the basin and reaches the 12-mile dumpsite 12% of the time. Despite the high contaminant loadings in the estuarine plume, the plume's influence on bottom habitat quality in the Christiaensen Basin could be minor or major, depending on the location of the plume and the efficiency of its delivery of particle-associated pollutants to the basin seabed. Similarly, the delivery efficiency of sewage sludge and its contaminants to sediments in the Christiaensen Basin was not well understood. Sewage sludge contains about 5% solids, is diluted several-thousand fold shortly after release from the barge (Calloway et al., 1976; Proni, 1976; Draxler, 1979), and is rapidly dispersed over an area much greater than the nominal size of the dumpsite. The fate of the dredged material, the largest source of contaminants to the apex, seems more certain (Fig. 2). It is generally believed that most of the material remains at the site (Dayal et al., 1981). Some of the contaminated fine-grained sediments at the dredge site may be resuspended and dispersed by storms (Young et al., 1985). How much of this finds its way into the adjacent Christiaensen Basin is unknown.

The studies of seabed sediments reported in this volume had several objectives. The first was to evaluate twelve hypotheses formulated prior to the initiation of

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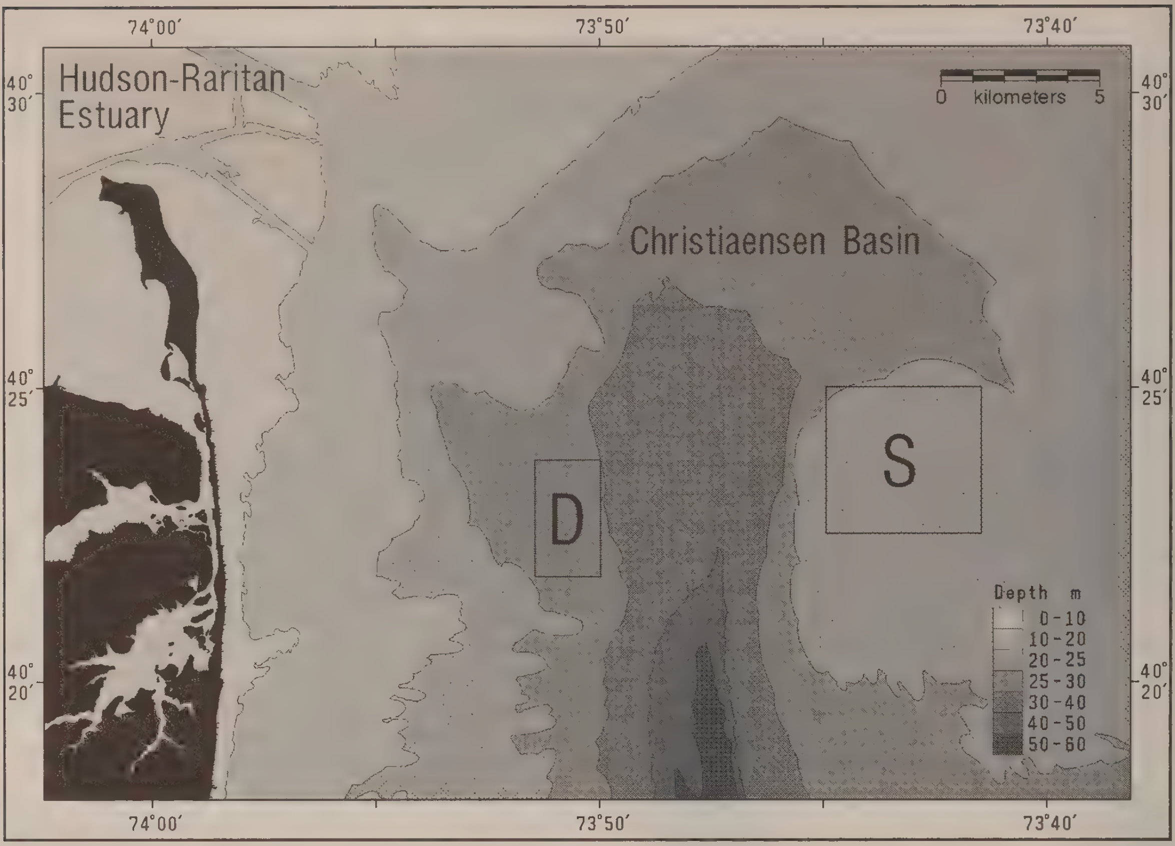

Figure 1

The locations of the 12-mile sewage sludge (S) and dredge material (D) dumpsites, the Christiaensen Basin (depths >25m), the mouth of the Hudson–Raritan Estuary, and the bathymetry of the inner New York Bight.

the study (EPD [Environmental Processes Division], 1988). These dealt with anticipated changes in sediment biogeochemical processes and contamination following the cessation of sludge dumping and were evaluated by conducting statistical comparisons of data from the dumping and post-dumping periods.

The second objective was to characterize the temporal and spatial changes in the contaminants and processes and to resolve abatement trends from seasonal and other sources of variation. Investigators were most optimistic about being able to quantify changes in sediments resulting from the closure of the site because 1) historical data were particularly abundant and of high quality, and 2) the pollution signal in the sediments was exceptionally strong from many decades of dumping.

Another objective was to resolve sources of sediment contaminants in the Christiaensen Basin. What is the relative importance of contaminants from sludge disposal activities, dredge disposal activities, and the contaminants brought into the area from the estuarine plume? It was uncertain whether contaminants from the estuarine plume and dredge disposal would mask the responses from cessation of sludge dumping.

The last major objective was to develop empirical models relating sediment processes to carbon loading and to benthic community responses. The unprecedented monthly sampling frequency and the multidisciplinary nature of the study made this possible.

We expected changes in sediment biogeochemical processes and that decreased contamination would accompany reductions in sludge inputs to the apex, but many of the details of the response were uncertain or unknown. A few of the questions investigators had in mind in 1985 and 1986 were the following:

> How long after cessation would change be evident?
>
> How fast would the rates of "recovery" be (years, decades)?
>
> How much of the NYB would be affected?
>
> How deep in the sediment column would changes be apparent?

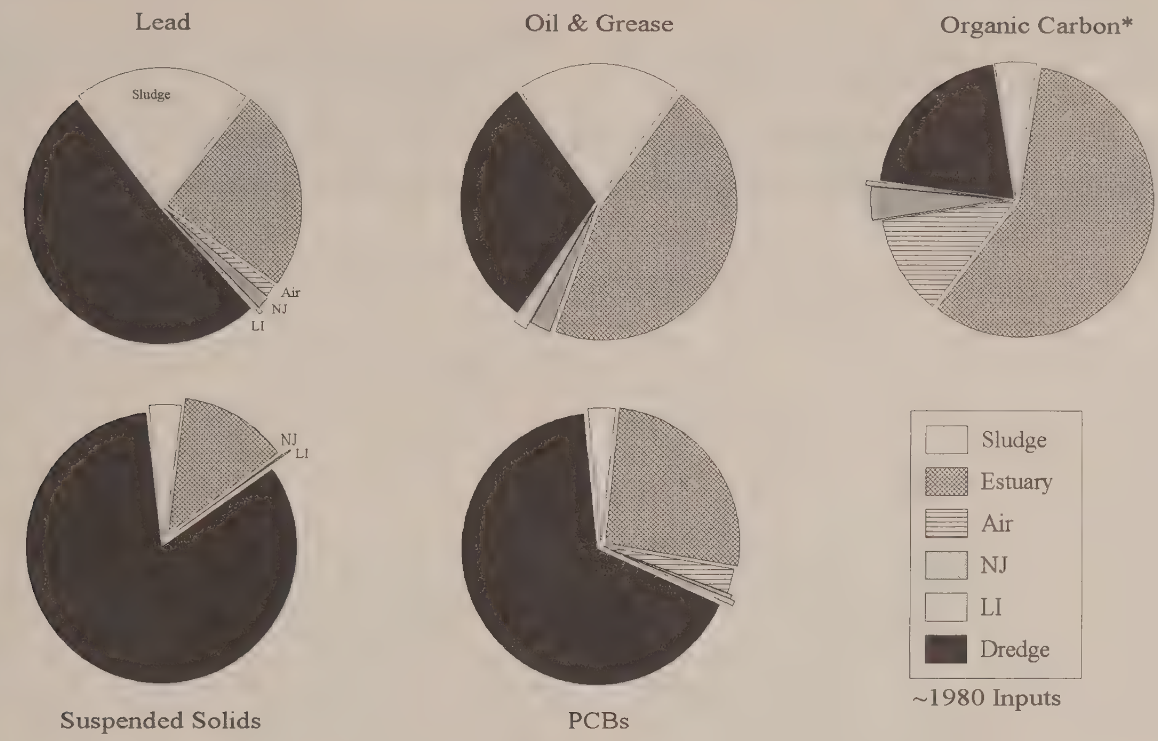

Figure 2
The distribution (%) of annual loads of various pollutants to the New York Bight apex. Data for lead, oil and grease, suspended solids, and PCB's are from Stanford and Young (1988). The distribution of organic carbon loads includes the entire New York Bight area (Mueller et al. 1976).

Could effects from sludge abatement be resolved easily from seasonal variation?

How would bathymetry, sediment type, and distance from the dumpsite affect the responses?

Would the rates of change be similar for a number of indices of sewage pollution?

What happens when one of the major sources of pollution to the NY Bight ceases, as happened in December 1987?

How significant was dumping at the 12-mile site to sediment processes in the apex and in the Christiaensen Basin?

The following papers address these questions.

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Reductions in Sediment Metal Contamination in the New York Bight Apex with the Cessation of Sewage Sludge Dumping

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ABSTRACT

Metal concentrations in sediments of the New York Bight apex were monitored to determine whether changes in sediment contamination would accompany the phase-out and cessation of sewage sludge dumping at a site 12 miles east of the New Jersey coast that had been in use since 1924. Sediments were collected at 25 stations during a study conducted between July 1986 and November 1989. At a highly contaminated station nearest the dumpsite (approximately 1.5 km to the west), metal levels in surface sediment decreased by a factor of approximately two within one month of complete cessation of sludge dumping. However, post-cessation concentrations in surface sediment remained elevated beyond pre-industrial levels; concentrations in buried sediment were similar to post-cessation surface sediment levels. Metal concentrations in surface and buried sediment from a moderately contaminated station (approximately 4 km northwest) and a reference station (approximately 11 km south) did not change significantly with cessation. In the study area as a whole, the spatial extent and degree of surface sediment pollution decreased during the phasing out and after the cessation of sludge dumping.

Introduction

Sewage sludge has been dumped in the New York Bight at a site 12 miles east of the New Jersey coast since 1924. Because of the presence of this and other dumpsites in the area, contaminant levels have been monitored in sediment and biota of the Bight, and more extensively in the Bight apex, since the late 1960's as a part of numerous studies by the National Oceanic and Atmospheric Administration (NOAA), the U. S. Army Corps of Engineers (COE), the Environmental Protection Agency (EPA) and others (Carmody et al., 1973; Reid et al., 1982; Zdanowicz, 1991; Boehm et al.[1]). These studies have shown that high concentrations of organic and inorganic contaminants have consistently been found in and around the dredged spoils and sewage sludge dumpsites, the Christiaensen Basin, and the upper reaches of the Hudson Shelf Valley (Fig. 1).

In 1985, EPA mandated that use of the 12-mile site be terminated and that all sewage sludge be disposed of at the 106-mile dumpsite, a deep water disposal area 106 miles east of Cape May, NJ. Sludge dumping at the 12-mile dumpsite was phased out beginning in March 1986 and was discontinued by December 1987. The closure of the site provided an opportunity to study the changes in levels of seabed contamination that might occur with reductions in sludge inputs, as well as an opportunity to monitor anticipated improvements in benthic habitat quality.

Methods

The study area, monitoring schedule, and sampling scheme have been described in detail (EPD [Environmental Processes Division], 1988; Pikanowski, 1995) and only a brief summary is provided here. The study consisted of two sampling strategies, denoted "Broadscale" and "Replicate." During "Broadscale" sur-

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[1] Boehm, P. D., W. Steinhauer, and J. Brown. 1984. Organic pollutant biogeochemical studies: Northeast United States marine environment. Final report for contract NA-83-FA-C-00022, Northeast Fisheries Science Center, National Marine Fisheries Service, NOAA, Sandy Hook Laboratory, Highlands, NJ, 61 p.

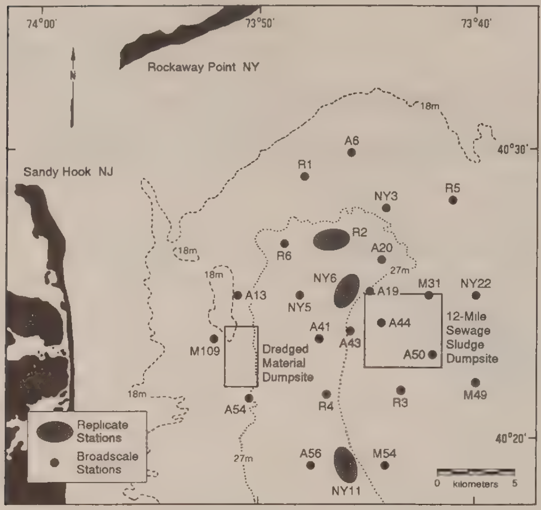

Figure 1

Locations of stations sampled during the 12-Mile Dumpsite Study.

veys, 22 stations situated throughout the Bight apex (Fig. 1) were sampled every other month; one sediment sample was collected at each station during each survey. The results of those surveys will be the topic of a separate report. During replicate surveys, multiple sediment samples were collected monthly at three stations (NY6, R2, and NY11; see Fig. 1). The results of the replicate surveys are the subject of this report.

Sediment samples were collected using a $0.1\ m^2$ Smith-MacIntyre grab equipped with rubber flaps over the doors to prevent disturbance of the sediment surface during retrieval of the sampler. Core subsamples were removed from the grab and stored frozen, upright. In the laboratory, cores were thawed and sectioned into 1 cm thick layers, which were dried, homogenized, and stored in polyethylene containers until analyzed.

A detailed description of the analytical methods is given in Zdanowicz et al.[2] Briefly, a strong acid leach (concentrated HNO_3, followed by dilute aqua regia) was used to extract chromium (Cr), copper (Cu), nickel (Ni), lead (Pb), zinc (Zn), and iron (Fe) from the dried sediment samples. This type of extraction has been shown to produce good recoveries for metals in polluted sediments (Förster and Salomons, 1980). Cr, Cu, Ni, Pb, and Zn are contaminants generally found in elevated concentrations in sewage sludge, while Fe, although also present in sludge, is generally associated with the fine fractions of a sediment and can be used to normalize trace element data (Trefry and Presley, 1976; Ackerman, 1980). Extracts were analyzed by flame atomic absorption spectrophotometry. Trace metal concentrations are expressed as mean values in ppm (mg/kg) dry weight, while iron values are in percent dry weight.

One core from each of three grabs collected monthly from three stations (NY6, R2, and NY11) was processed. Two sediment strata were analyzed, the surface (0–1 cm) layer and a buried (4–5 cm) layer. Surface sediment samples were used to assess changes in the most recent inputs of metals at each station and, in conjunction with buried sediment samples, to provide information on the vertical distributions of contaminants in the sediment column. Vertical chemical distributions provide information on the depositional history of a site.

Trace metal concentrations in sediment are naturally variable. They depend on the grain size composition of the sediments, since trace metals are associated primarily with fine sediment fractions, such as clays and fine silts (Förstner, 1983; Windom et al., 1989). In the New York Bight, grain size composition of sediments varies widely (Stubblefield et al., 1977). Trace metal data from different sediment types can be made more directly comparable by normalizing trace element concentrations to some other sediment variable associated with the fine sediment fraction. Those most commonly used are certain fine fractions (deGroot et al., 1976; Förstner, 1982) or elements that are major sediment constituents associated with fine sediment, usually iron (Trefry and Presley, 1976; Ackerman, 1980) or aluminum (Bertine and Goldberg, 1977; Goldberg et al., 1979; Klinkhammer and Bender, 1981; Finney and Huh, 1989). Normalization can also provide information on contaminant sources (Anders, 1972; Keeler and Samson, 1989; Poulton, 1989; Zdanowicz, 1991) and, if suitable reference data are available that describe metal levels in uncontaminated sediment (Zdanowicz and Finneran[3]), enrichment factors (EF's) can be derived from normalized data and used to assess the extent of contamination of a sediment. For this study, pre-industrial New York Harbor sediment (Williams et al., 1978) was used to estimate the degree of contamination of New York Bight apex sediment. Using a modification of the formula given by Kemp et al. (1976), EF's were computed as

$$EF = (\text{metal}/\text{iron})_{NYB} / (\text{metal}/\text{iron})_{PIH}$$

where "NYB" indicates sediments from this study and "PIH" indicates the pre-industrial New York Harbor sediment.

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[2] Zdanowicz, V. S., S. L. Cunneff, and T. W. Finneran. In prep. 1995. Metal concentrations in sediments of the New York Bight apex before and after cessation of sewage sludge dumping. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-F/NEC, Woods Hole, MA.

[3] Zdanowicz, V. S., and T. W. Finneran. 1995. Pre-industrial metal concentrations in New York Harbor sediment. In prep.

As an illustration of the utility of EF's, metal concentrations determined in this laboratory, metal/iron ratios, and contaminant metal EF's from selected reference sediments are shown in Table 1. EPA 2988 is a freeze-dried, municipal sewage sludge (obtained from EPA, Environmental Monitoring and Support Laboratory, Cincinnati, Ohio), used in this study to represent a "typical" sludge. RB is fine, moderately contaminated, surface sediment from Raritan Bay collected during the mid 1980's (Zdanowicz and Gadbois, 1990), included here because this estuary is a possible source of solids to the Bight apex. PIH is the modern, pre-industrial New York Harbor sediment mentioned above. The computed EF's clearly demonstrate the high degree of contamination of sludge and the reduced degree of contamination of Raritan Bay sediment.

Statistical procedures for investigating possible effects of the cessation of sludge dumping were based on a Before-After-Reference-Polluted design (Pikanowski, 1995). The experimental "treatment" was the removal of sludge inputs. "Before" signifies the period before cessation; "After" signifies the period after dumping activities were terminated (the cessation period). There were 18 surveys during the "Before" period and 23 during the "After" period. The three replicate stations were each presumed contaminated to various degrees (EPD, 1988), based on prior studies. Station NY11, presumed least contaminated, was used as the reference station. The "most polluted" station was NY6; station R2 was intermediate in contamination. The "raw" variables were metal concentrations, trace metal/iron ratios, and trace metal EF's. The statistical variables were the differences between means (DM's) of the raw variables at a polluted station (NY6 or R2) and the reference station (NY11) for each survey. Statistical analyses were performed on both the DM's and the raw variables.

Results

Results of metal analyses of surface and buried sediment samples are shown in Figures 2–4. Vertical lines in the figures delineate different dumping periods. July 1986 to June 1987 (left) will be referred to as the "70%" period, because during this time approximately 70% of all local sludge dumped at sea was dumped at the 12-mile site. The period between June 1987 and January 1988 (center) will be referred to as the "phase-out" period, because during that time the amount of sludge dumped at the 12-mile site was reduced stepwise from approximately 70% to approximately 30% of all local sludge dumped at sea. January 1988 to November 1989 (right) will be referred to as the "cessation" period, because use of the site ended in December 1987.

Table 1
Composition of reference sediments.

Mean concentrations (ppm, dry weight)

		Cr	Cu	Ni	Pb	Zn	Fe(%)
EPA 2988[1]	(n = 3)	220	1060	211	556	1140	1.92
RB[2]	(n = 12)	188	182	40	169	431	4.55
PIH[3]	(n = 6)	62	14	23	13	74	3.30

Trace metal/iron ratios (ppm / %)

	Cr/Fe	Cu/Fe	Ni/Fe	Pb/Fe	Zn/Fe
EPA 2988	115	555	110	290	594
RB	41	40	9	37	95
PIH	19	4	7	4	22

Enrichment factors (see Materials and Methods for computation)

	Cr	Cu	Ni	Pb	Zn
EPA 2988	6.1	139	16	72.5	27
RB	2.2	10	1.3	9.3	4.3

[1] EPA 2988 — freeze-dried, municipal sludge.

[2] RB — fine surface sediment from Raritan Bay.

[3] PIH — pre-industrial sediment from the 75–80 cm layer of a core collected at m.p. 0 in New York Harbor (see Williams et al, 1978).

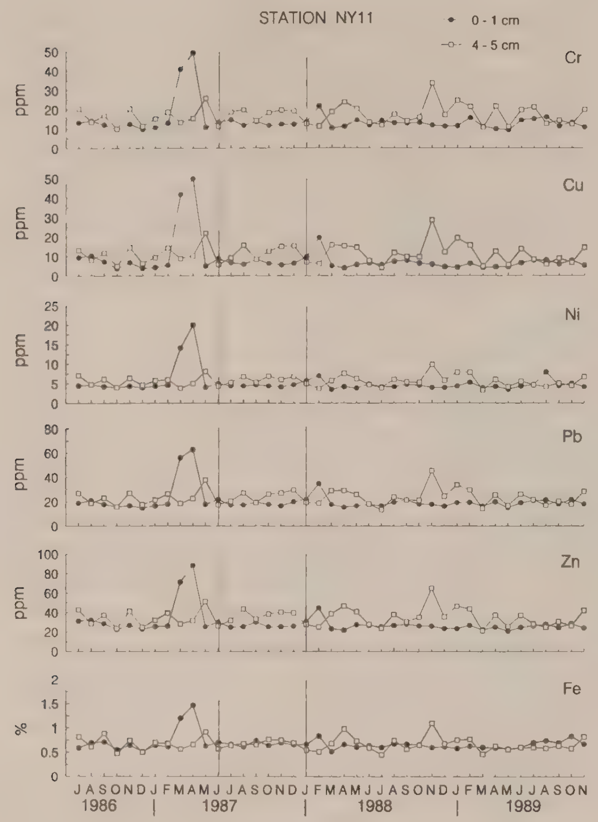

Figure 2

Metal concentrations in surface and buried sediment at station NY11 (ppm dry weight). Vertical lines delineate the 70% dumping period (left), the phase-out period (center), and the cessation period (right).

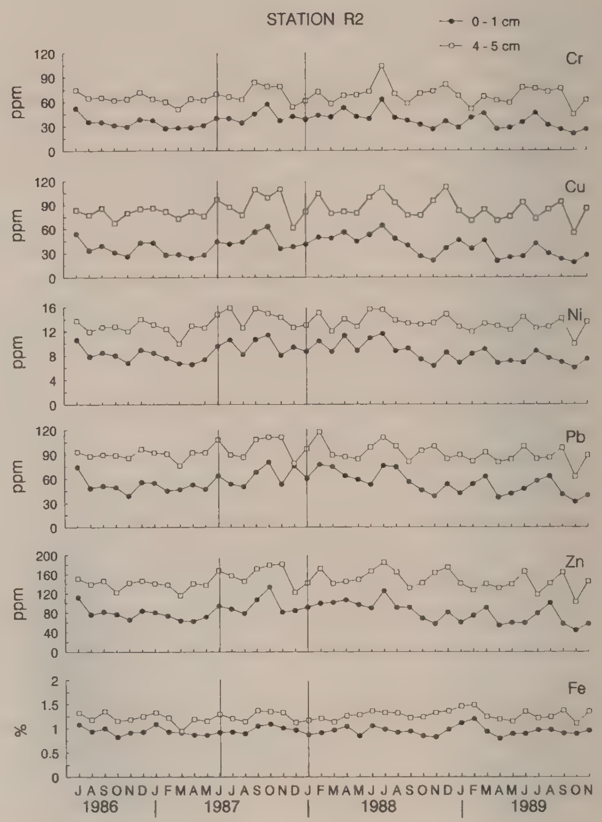

Figure 3

Metal concentrations in surface and buried sediment at station R2 (ppm dry weight). Vertical lines delineate the 70% dumping period (left), the phase-out period (center), and the cessation period (right).

Station NY11

Among the three Replicate stations, the lowest metal concentrations were found at NY11, the reference station (Fig. 2, Table 2). Metal levels were similarly low in the surface and buried sediment layers and generally comparable to metal levels in clean sediment, although lead concentrations were moderately elevated. Metal levels remained relatively constant throughout the study and were consistently lower than those at stations NY6 and R2. The increases in contaminant metal concentrations observed in March and April 1987 in surface sediment and in November 1988 in buried sediment were accompanied by increases in iron concentrations, suggesting that increases in contaminant concentrations were probably due to the presence of higher proportions of fine sediment in those samples. No statistically significant differences were found among metal concentrations, metal/iron ratios, or metal EF's observed during the dumping and cessation periods in either sediment layer. Statistical analysis of the DM's yielded the same results.

Station R2

Metal concentrations in R2 sediment were intermediate between those at stations NY6 and NY11. Contaminant concentrations in surface sediment at station R2 (Fig. 3, Table 2) were greater than those in surface sediment at NY11 by factors of roughly 1.5 (iron) to 5 (copper) both before and after cessation. Metal levels in buried sediment at R2 were greater than those in buried sediment at NY11 by factors of about 2 (iron) to 7 (copper). At R2, unlike both NY6 and NY11, metal levels in the buried layer were distinctly higher than in the surface layer. This situation was previously observed in sediment samples collected in 1982 and 1983 from stations near R2; a fairly smooth gradient was observed, with metal concentrations increasing with sediment

Table 2

Comparison of replicate station results. B = mean of all observations before cessation; A = mean of all observations after cessation; * = significant difference (α <0.0005); ** = significant difference (α <0.005).

Sediment Depth		0–1 cm			4–5 cm		
Station		NY6	R2	NY11	NY6	R2	NY11
Cr (ppm)	B	163	37.2	15.8	97.8	66.2	16.7
	A	96.5*	36.7	12.9	103	68.0	17.3
Cu (ppm)	B	312	38.6	10.9	164	84.1	11.4
	A	151*	37.5	6.74	171	85.0	11.3
Ni (ppm)	B	29.5	8.58	5.81	18.6	13.2	5.71
	A	16.9*	8.32	4.67	18.9	13.3	5.64
Pb (ppm)	B	272	56.0	22.5	181	93.0	23.4
	A	153*	54.3	19.3	181	90.8	23.6
Zn (ppm)	B	532	83.9	32.7	289	147	35.2
	A	232*	79.6	26.4	288	147	34.9
Fe (%)	B	1.00	0.953	0.720	0.720	1.22	0.682
	A	0.769*	0.945	0.653	0.742	1.28	0.664
Cr/Fe	B	157	38.3	19.8	128	53.7	23.7
	A	123**	38.6	19.8	133	53.2	25.7
Cu/Fe	B	296	39.1	11.6	211	68.0	15.3
	A	190*	38.9	10.0	220	66.2	15.8
Ni/Fe	B	28.6	8.92	7.19	25.1	10.8	8.25
	A	22.0*	8.81	7.11	25.0	10.4	8.44
Pb/Fe	B	269	57.8	29.0	243	75.6	33.4
	A	198*	57.4	29.5	243	71.1	35.0
Zn/Fe	B	504	86.6	42.9	375	119	50.9
	A	293*	83.8	40.5	371	115	52.0
Cr EF	B	7.8	1.9	1.0	6.4	2.7	1.2
	A	6.1**	1.9	1.0	6.6	2.7	1.3
Cu EF	B	74	9.8	2.9	53	17	3.8
	A	48*	9.7	2.5	55	17	3.9
Ni EF	B	4.1	1.3	1.0	3.6	1.5	1.2
	A	3.1*	1.3	1.0	3.6	1.5	1.2
Pb EF	B	67	14	7.2	61	19	8.4
	A	49*	14	7.4	61	18	8.7
Zn EF	B	23	3.9	2.0	17	5.4	2.3
	A	13*	3.8	1.8	17	5.2	2.4

depth (Zdanowicz, unpublished data). This may be due to higher proportions of fine sediment in buried sediment layers in this area, as evidenced by higher iron levels in samples of buried sediment. Concentrations of all six metals in both sediment layers were fairly consistent over the course of the study and there were no statistically significant differences among metal concentrations, metal/iron ratios, or metal EF's observed during the dumping and cessation periods in either layer. Statistical analysis of the DM's produced similar results.

Station NY6

The highest metal concentrations observed at the Replicate stations were found in samples of surface sediment from station NY6 (Fig. 4, Table 2) collected during the 70% and phase-out periods. Levels of all five contaminant metals were highly elevated relative to the reference station (NY11): iron levels were only slightly elevated. For example, before cessation, copper in surface sediment was almost 30-fold greater at NY6 than at NY11, whereas after cessation copper was roughly 22-fold higher. Surface sediment metal concentrations at NY6 during the two dumping periods (the 70% and phase-out periods) were higher than those in the buried layer by a factor of approximately 1.5 (iron) to 2 (copper). After cessation, metal levels in surface sediment had generally declined to levels similar to those found in the buried layer, although both sediment layers remained contaminated. Thus, both strata were contaminated throughout the course of the study, with

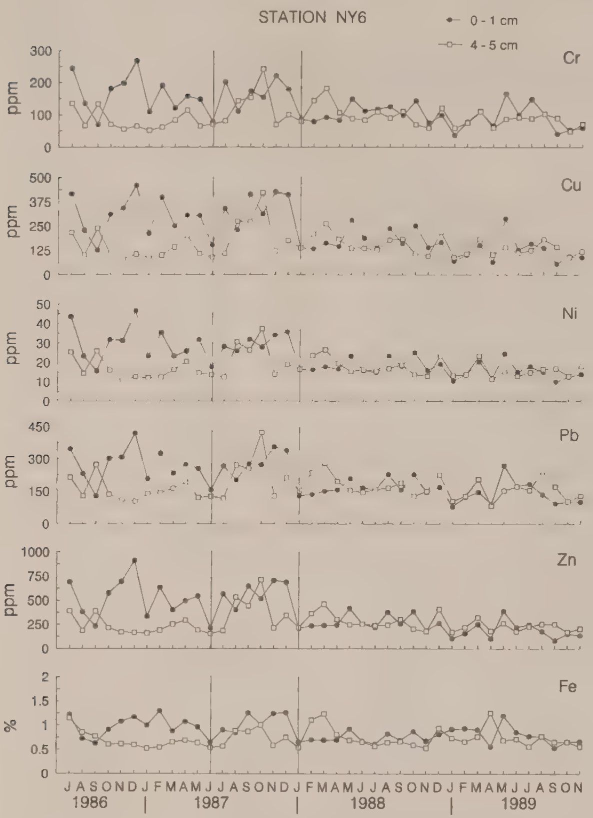

Figure 4

Metal concentrations in surface and buried sediment at station NY6 (ppm dry weight). Vertical lines delineate the 70% dumping period (left), the phase-out period (center), and the cessation period (right).

surface sediment more highly contaminated than buried sediment during dumping.

Levels of all six metals were significantly (<0.005) lower in surface sediment at NY6 after cessation (Table 2). Metal/iron ratios and contaminant metal EF's also were significantly lower in surface sediment after cessation. No significant differences were observed for buried sediment. Statistical treatment of the DM's produced the same results. In a separate statistical analysis, the 70%, phase-out, and cessation periods were compared (ANOVA, =0.05). Results indicated that metal concentrations in surface sediment during the two dumping periods were indistinguishable and that concentrations after cessation were significantly lower than during either the 70% or phase-out period. Again, no significant differences were observed for buried sediment.

Discussion

Metal concentrations indicate that both surface and buried sediment at station NY6 were highly contaminated with sewage sludge. Surface sediment EF's at NY6 were the greatest of the study (Fig. 5, Table 2), ranging from 4.1 for nickel to 74 for copper during sludge dumping. After cessation, EF's decreased significantly, but remained elevated, ranging from 3.1 for nickel to 49 for lead. In buried sediments, EF's both before and after cessation were similar to surface EF's after cessation, ranging from 3.6 for nickel to 61 for lead, and did not change significantly after cessation. Thus, with the cessation of sludge dumping, the most contaminated inputs to the Bight apex were eliminated and, with them, the presence of a highly contaminated layer of material at the sediment surface. However, a reservoir of contaminants remains in NY6 sediment, having been incorporated into the seabed during the years of sludge dumping. That metal concentrations and EF's during the 70% and phase-out periods were indistinguishable is of particular interest. This finding suggests that dumping as little as 0.7 million t per quarter year of all locally generated sludge at the 12-mile site might be sufficient to contaminate the benthic habitat at NY6.

A combination of processes is likely responsible for the observed distributions. During a dumping event, sludge, which is typically 95% water, is rapidly diluted by a factor of 10^4 to 10^6 (Draxler, 1979; Jenkins and Gibbs, 1988). Sludge solids fall to the sea floor at varying rates, depending on their size, density, and other factors. Simultaneously, they may be transported by local currents while still in the water column. Station NY6 is situated in a depression (the Christiaensen Basin), adjacent to the 12-mile site, allowing some sludge solids to settle. Once on the seabed, solids can become mixed into the sediment column through mechanical sorting and bioturbation by resident benthic fauna. Solids can also be resuspended through bioturbation and/or storm-induced currents, and transported away from the site. Resuspension, transport and redeposition can also introduce solids from other areas to NY6. Other investigations undertaken during this study show that the hydrography of the Bight apex is complex (Mountain and Arlen, 1995) and that both upwelling and downwelling events occur in the vicinity of the Hudson Shelf Valley and Christiaensen Basin (Manning, 1995). These events can result in sediment transport (Davis et al., 1995). In addition, a diverse and abundant benthic community is present throughout the study area (Reid et al., 1995). Of particular importance is the fact that portions of the Christiaensen Basin are areas of net deposition (Bopp et al., 1995), 4 mm/yr at NY6 since 1954.

Thus, the observed contaminant distributions and enrichments are the results of a combination of several complex processes. During many years of dumping, pulsed sludge inputs deposited contaminated solids on

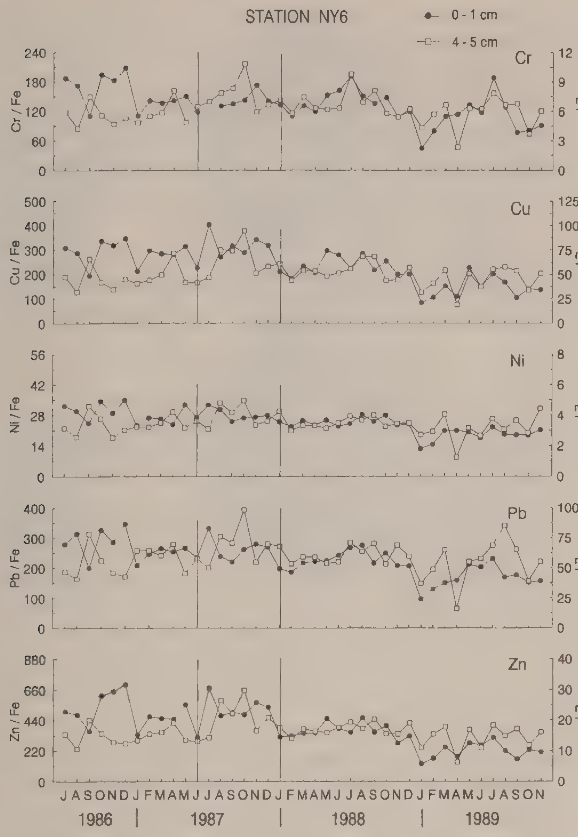

Figure 5
Metal/iron ratios and enrichment factors in surface and buried sediment at station NY6. Vertical lines delineate the 70% dumping period (left), the phase-out period (center), and the cessation period (right).

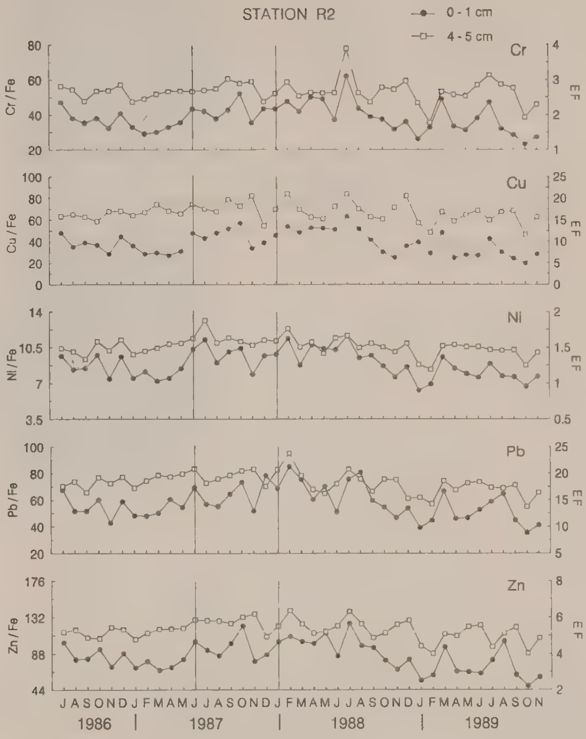

Figure 6
Metal/iron ratios and enrichment factors in surface and buried sediment at station R2. Vertical lines delineate the 70% dumping period (left), the phase-out period (center), and the cessation period (right).

the sediment surface, causing highly elevated contaminant concentrations and EF's. Since these materials were unconsolidated and fairly easily transported, their effects disappeared rapidly following the cessation of dumping. However, each occurrence of sludge settling on the seabed may have resulted in the incorporation of some contaminated material into the sediment column, such that, even after the removal of sludge inputs from the system, a contaminant reservoir at least 20 cm deep remains (Bopp et al., 1995).

Station R2 presents a situation quite different from that of NY6 (Fig. 6). No statistically significant changes occurred in metal concentrations or metal EF's with the cessation of sludge dumping. In fact, metal concentrations and EF's from the Before and After cessation periods were remarkably constant (Table 2). Numerically, an EF > 1 indicates the presence of excess metal, but, due to sample and measurement variability, a more conservative criterion is to use an EF≥2 to indicate enrichment. On this basis, chromium and nickel were not enriched in R2 surface sediment, while copper, lead, and zinc were enriched. EF's in surface sediment were very similar to EF's of Raritan Bay sediment (Table 1), suggesting the possibility that material from that estuary is present in R2 surface sediment. It seems unlikely, however, that there could be an appreciable quantity of Raritan Bay material in R2 sediment, given the distance from that estuary to R2, and the fact that the Hudson plume hugs the New Jersey coast most of the time (Fedosh and Munday, 1982). In buried sediment, although the nickel EF was similar to that in the surface sediment, EF's for the other metals were higher. Thus, material present in buried sediment at R2 is more contaminated than material found in surface sediment. It is possible that a layer of contaminated material was deposited there earlier and is slowly being winnowed out. This seems more likely than preferential burial of contaminated material. If illegal dumping in

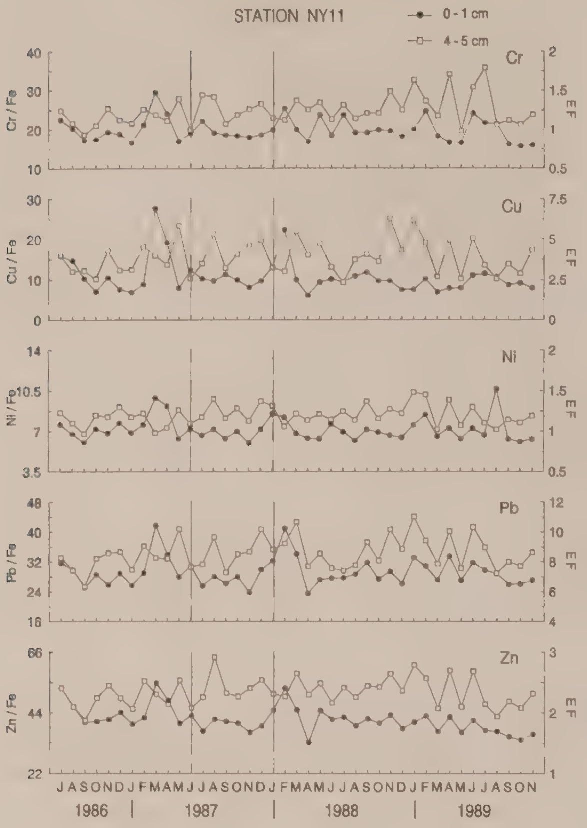

Figure 7

Metal/iron ratios and enrichment factors in surface and buried sediment at station NY11. Vertical lines delineate the 70% dumping period (left), the phase-out period (center), and the cessation period (right).

the northern Christiaensen Basin of sludge or dredged spoils can be ruled out, the source of this contaminated material is probably sludge dumped at the 12-mile site, since alternate sources, including the Hudson-Raritan outflow and the dredged spoils dumpsite, can be discounted with reasonable certainty. That is, Raritan Bay sediments simply are not enriched with these metals to this extent (Table 1). And, although dredged spoils may be highly contaminated with metals, depending on the area being dredged, they are not easily transported after settling at the dumpsite. Most of the mass dumped at the dredged spoils dumpsite has remained within its confines (Dayal et al., 1981).

At station NY11 (Fig. 7), as at R2, no significant changes occurred in metal concentrations or metal EF's in either sediment layer with the cessation of sludge dumping. Metal concentrations and EF's from the Before and After cessation periods were surprisingly constant in both sediment layers (Table 2). Little or no enrichment in chromium, nickel, and zinc was evident in either surface or buried sediments. Copper was minimally enriched in surface sediment and slightly more enriched in buried sediment, while lead enrichment was moderate and comparable in both layers (Table 2). The presence of *Clostridium perfringens* spores indicates sludge contamination at this station (O'Reilly et al., 1995), but the source or sources and mode of incorporation into the sediment are uncertain. Sludge may have been deposited directly at this station or transported from the dumpsite, or sludge-contaminated material from the dredged spoils dumpsite or Hudson Shelf Valley could have been deposited there during one or more transport events. Of these possibilities, the last is more probable, since the Hudson Shelf Valley is known to be a conduit for contaminated materials between the Christiaensen Basin and the mid Shelf. The strong similarities between metal concentrations in the two sediment layers and EF's in the two layers suggest that the same source contributed the enriched material found in both layers, that the composition of that material was fairly constant, and that the net rate of input, whether constant or pulsed, was slower than the rate of mixing into the sediment column. These considerations also favor the Hudson Shelf Valley as the source of contaminated materials to NY11.

In order to place these results in a broader areal context and provide additional evidence of changes in sediment contamination that occurred with the cessation of sludge dumping, metal EF's of samples collected during "Broadscale" cruises were contoured. Figure 8 shows four contour plots of lead EF's in surface sediment from the study area. The surveys shown were conducted during August 1986 (70% dumping period, first "Broadscale" cruise), December 1987 (phase-out period, 30% of all local sludge dumped at the 12-mile site), February 1988 (first cruise after cessation), and August 1989 (last cruise of the study). The total area contoured covers 350 km^2. The "most polluted" portion of the apex was where lead EF's exceeded 20. This area included NY6, where, before cessation, lead EF's were >50. Of note are the steep pollution gradients and the high degree of heterogeneity in the study area.

The portion of the study area with EF >20 decreased from about 17% (60 km^2) of the total in August 1986 (average during the 70% dumping period) to about 8% (28 km^2) in August 1989, indicating a reduction in pollution of Bight apex surface sediment with time. Variations in the percent of the study area with lead EF >20 are plotted versus time in Fig. 9, along with the volume of sludge dumped per quarter year; contour plots of the area with lead EF >20 during the four surveys shown in Fig. 8 are inset. During the first four surveys in the 70% dumping period, an average of

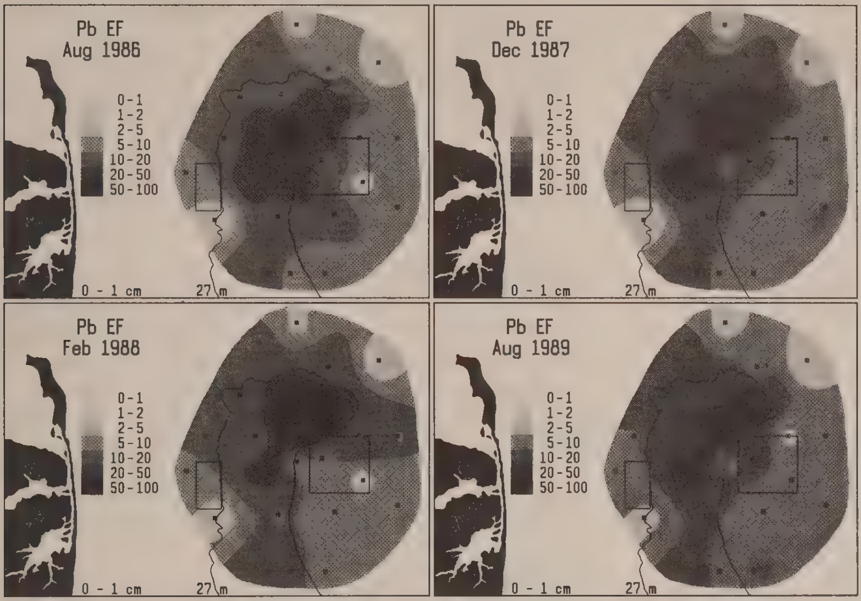

Figure 8
Lead EF's in surface sediment during selected broadscale surveys.

roughly 23% (81 km^2) of the study area was highly polluted with lead, while during the last two surveys in that period and the three surveys in the phase-out period, an average of approximately 13% (46 km^2) of the area was highly polluted with lead. After cessation, the percentage of highly polluted area decreased with time. A linear regression model produced the best fit to the observed decrease and indicated that the "most polluted" area appears to be decreasing at a rate of 0.92 km^2 per month. By extrapolation, this area should completely disappear within 48 months of cessation (by January 1992). Similar analyses were performed using data on the other contaminant metals. Time scales for recovery from the "worst" pollution (highest degree of enrichment) were all less then 48 months. The recovery mechanism includes such processes as resuspension induced by bioturbation and events related to weather, tides, and currents, followed by transport. When these resuspension events occur during periods of no net export from the Christiaensen Basin, the contaminants will merely be redistributed without any reduction in total contaminant load. However, when the contaminated solids are resuspended and transported during periods of net export from the Basin, the net contaminant load in Basin sediments will be reduced; these events coupled with downwelling events would likely result in the deposition of contaminated material in the Hudson Shelf Valley.

These projections apply only to the surface layer of sediment. The future of the buried contaminant reservoir cannot be predicted and one can only speculate regarding its fate. If dredged spoils remain within the boundaries of that dumpsite and the Hudson plume contributes negligible amounts of contaminated solids to the Bight apex, and if no new anthropogenic sources of contamination materialize, the distributions of contaminants in apex sediments will be governed solely by natural processes. Contaminant levels in areas of no net accumulation would be expected to decrease, due to resuspension and transport of contaminated sediment, resulting in an overall reduction of pollutant loads. However, if contamination is present in these areas at depths below the zone of appreciable bioturbation or resuspension, a deep layer of contaminated sediment would remain in place, covered with cleaner material. In areas of net deposition, contaminant levels in sediments should also decrease, since newer materials deposited there should be cleaner than the sludge-contaminated materials deposited during dumping. This, too, would produce a buried layer of contaminated sediment covered with clean material. Thus, it is possible that sediment in buried strata might remain contaminated.

Conclusions

Of particular importance are the magnitude and statistical significance of the decreases in surface sediment metal levels at NY6 since the cessation of sludge dump-

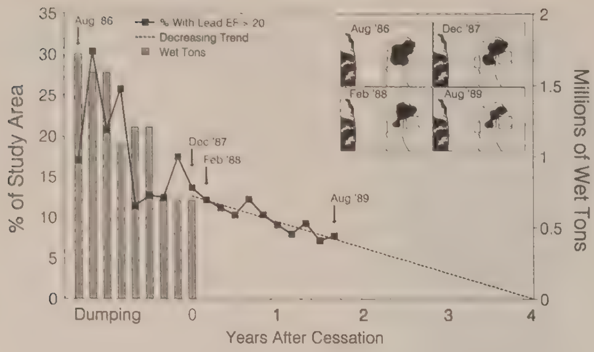

Figure 9

Amount of sludge dumped and the observed and projected percentages of the study area with lead enrichment factor >20. Insets are contour plots of portions of study area with Pb EF >20, derived from Fig. 8.

ing. From the 70% dumping period to the phase-out period, there was a two-fold decrease in the volume of sludge dumped at the 12-mile site, yet surface sediment metal concentrations during those periods were statistically indistinguishable—only after cessation were contaminant concentrations significantly lower. Before cessation, metal concentrations did not change with the decrease in volume of sludge dumped because the sludges dumped presumably were contaminated to a similar extent, and, as long as a sufficient quantity of sludge was dumped to cause accumulation on the surface of NY6 sediment, metal concentrations remained elevated. This situation would be expected to persist until the volume of sludge dumped were reduced to some level below which appreciable accumulation does not occur. What this "critical" level might be, however, cannot be computed from these data.

In addition, the observed decreases occurred in January 1988, within one month of cessation of dumping, indicating rapid response to the removal of contaminant inputs. Of equal importance is that, even after the elimination of the most polluted inputs to the system, sediment at NY6 remained highly contaminated, apparently due to the incorporation into the sediment column of sludge derived materials introduced during dumping.

The percentage of the overall study area where surface sediment exhibited the "worst pollution" decreased with the reduction in sludge dumping volume and appeared to be declining gradually after cessation. Based on the observed rates of decline, it is estimated that the worst pollution will have been eliminated within four years of the cessation of dumping (by January 1992).

Also of interest are findings at stations R2 and NY11. Sewage contamination is present at both stations, as evidenced by metal EF's. Metal concentrations and levels of enrichment at R2 were higher than at NY11 and were higher in buried sediment than in surface sediment, possibly due to a layer of contaminated material deposited earlier that is slowly being winnowed. At NY11, metal concentrations and enrichments were relatively constant and low, suggesting that this station may be indirectly influenced by sludge dumping through transport to this site of enriched material of sludge origin.

Acknowledgments

We thank R. Bopp for providing the PIH sample, P. Fournier for database management, E. Leimburg for data processing, and A. Draxler, J. O'Reilly, A. Studholme, and several anonymous reviewers for providing helpful comments. And a special thanks to all those who spent many days at sea during 41 months of field work.

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Audience Questions

Question: I understand you had a concern about changes in metal concentrations not being related to changes in mass-per-unit sediment but to mass-per-unit silt. Can you speculate how much the reduction was related to the change in silt-concentration chemistry versus the percent silt in established samples?

V. Zdanowicz: Our greatest change was seen in metal levels in surface sediment at NY6; there were almost no changes at the other replicate stations.

Question: Does this represent a change in the silt fraction? As we go from sand-silt to sand, the silt fraction is reduced approximately tenfold or something like that. I am interested in the change in chemistry as related to the change in the concentration of the silt fraction.

V. Zdanowicz: Actually I am not sure that question can be answered with these data. Iron is a good measurement to make. It is just one extra measurement on the same digest of the sediments, and we have used it as a substitute measurement for actual measurements of fine sediment fractions, which are much more elaborate and time consuming.

Question: Why did you choose a lead enrichment of 20 as the cut-off of the original samples?

V. Zdanowicz: That was based on results of a 1983 survey in the Bight apex (our annual summer survey) where we tried the same kind of techniques with normalization and enrichment factors. We were looking to separate the effects of dredged spoil dumping from sewage sludge dumping and the Hudson–Raritan plume. We found that we could separate contamination due to sludge and dredged spoil dumping only in sediment with enrichments of 20 or more.

Organic Contaminants in Sediments of the New York Bight Apex Associated with Sewage Sludge Dumping

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ABSTRACT

Higher levels of semi-volatile organic contaminants were found in sediments from a station located approximately 1.4 km northwest of the 12-mile sewage sludge dumpsite in the New York Bight apex as compared with two other stations: a moderately contaminated reference station 4.0 km north of the dumpsite and a reference station 11.0 km south-southwest of the dumpsite. Among organic contaminants, concentrations of polynuclear aromatic hydrocarbons generally exceeded those of chlorinated pesticides and polychlorinated biphenyls. Intersample variations in contaminant concentrations were apparent at all stations. In general, the mean concentrations of organic contaminants were lower in sediments collected in August 1989 than in those collected in August 1986 during dumping, but, given the limited number of samples, the difference was not statistically significant.

Introduction

Over the past half century, trace organic contaminants have entered the New York Bight apex (Fig. 1) by a variety of transport mechanisms including 1) dumping of sewage sludge from the New York metropolitan area at the 12-mile dumpsite, 2) dumping of dredged material from the New York Harbor at the 6-mile dredged material dumpsite, 3) incineration of wood at sea, 4) the Hudson–Raritan estuarine outflow containing contaminated suspended sediments (including those from storm water run-off), and 5) atmospheric deposition (Gross, 1976; Mueller et al., 1976; Stanford and Young, 1988; Gibson et al., 1979; O'Connor et al., 1982; Brown et al., 1985; Conner et al.[1]). Historical data (Boehm, 1982) have indicated that the sedimentary contaminant levels were highest in the sewage sludge accumulation area and at the dredged material dumpsite compared with sediments from adjacent locations. Moreover, contaminant levels in demersal biota in the sewage sludge accumulation area were elevated compared with concentrations reported outside the Bight (Boehm, 1982; Draxler et al., in press).

After over 60 years of dumping of sewage sludge at the 12-mile site (Fig. 1), a phased cessation of dumping began in March 1986, and by 31 December 1987, this site was closed for dumping (EPD, [Environmental Processes Division], 1988). The Northeast Fisheries Science Center (NEFSC) instituted a program to document changes in environmental and biological conditions in and around the 12-mile dumpsite and proposed several hypotheses regarding the recovery of the dumpsite after the dumping was phased out (EPD, 1988; Draxler et al., in press). The primary objective of this portion of the study was to test the hypothesis that levels of polychlorinated biphenyls (PCB's), selected as a marker for organic contaminants in sediment, would remain essentially unchanged (EPD, 1988).

Materials and Methods

Sediment Collection

Sediments were collected using a Smith McIntyre grab from the R/V *KYMA* during two complementary surveys—a replicate survey and a broadscale survey (EPD, 1988; Pikanowski, 1995). The top 1 cm layer of sediment was transferred to a dichloromethane-rinsed jar,

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[1] Conner, W. G., D. Aurand, M. Leslie, J. Slaughter, A. Amr, and F. I. Ravenscroft. 1979. Disposal of dredged material within the New York District, Vol. 1, Present practices and candidate alternatives. Mitre Corp., Metrek Division, McLean, VA, 384 p.

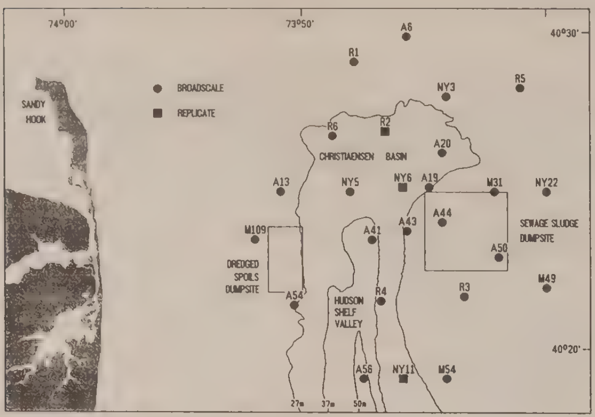

Figure 1

Replicate and broadscale station locations in and around the 6-mile dredge spoil dumpsite and the 12-mile sewage sludge dumpsite in the New York Bight apex.

with a Teflon-lined screw cap using a dichloromethane-rinsed stainless steel spoon, and was stored at –20°C.

In the replicate survey, a total of eight samples per station were collected from three replicate stations: NY6, R2, and NY11 (Fig. 1). Three of the stations were at the center of a sampling "ellipse" and five were at its perimeter; only samples from the three center stations were analyzed in this study. The replicate stations are bathymetrically similar, but they are different with respect to the levels of sewage sludge accumulation and the biological contaminant effects. Based on historical data on contaminant levels and benthic macrofauna, NY6 was expected to be the most heavily degraded station; R2, an organically-enriched station; and NY11, a relatively unpolluted station (EPD, 1988; Pikanowski, 1995). In the broadscale survey, one sample per station was collected from 25 stations in the New York Bight apex, except for stations NY6, R2, and N11, where three samples were collected during each broadscale survey (Fig. 1).

The present study focuses on the available results of organic analyses of sediments from the three replicate stations collected during four summer surveys: August 1986, August 1987, August 1988, and August 1989.

Analytical Method

Sediments were analyzed for 24 polynuclear aromatic hydrocarbons (PAH's), 24 polychlorinated biphenyls (PCB's) and 13 chlorinated pesticides (Tables 1, 2). The procedure of Krahn et al. (1988) was used for the simultaneous analysis of PAH's, PCB's, and chlorinated pesticides with one modification. We analyzed contaminants using a gas chromatography–mass selective detector (GC–MSD) instead of the gas chromatography–flame ionization detector (GC–FID) and gas chromatography-electron capture detector (GC–ECD) used in Krahn's procedure. Briefly, a mixture of sediment, sodium sulfate, activated copper, and methylene chloride was rolled in teflon-capped amber bottles for about 40 h in three steps. At each step, the mixture was centrifuged, methylene chloride extract was decanted, and fresh methylene chloride was added. Methylene chloride extract was precleaned by silica–alumina glass column chromatography. Biogenic and other interfering compounds were further removed by size-exclusion chromatography (Phenogel, 250 × 22.5 mm, 10 mm particle size, 100 Å pore size; Phenomenex, Torrance, California).

The analyte mixture was separated on two fused silica capillary columns connected end-to-end (PTE–5, 30 m × 0.25 mm ID, 0.25 mm film thickness; Supelco, Bellefonte, Pennsylvania) and was analyzed by a Hewlett Packard 5970B mass selective detector operated in select ion monitoring mode (Tables 1, 3). Analytes were identified by comparing retention times with reference standards (reference peak window = ±0.15 min) and quantified by the external standard method.

Appropriate internal standards were added to the sample at each step. A blank and a spiked blank were included with at least every 10 sediment samples. Recoveries of internal standards in blanks varied from 23 to 50% from analyte to analyte and experiment to experiment. Recoveries of PAH and PCB analytes spiked in reference sediments that were analyzed in this study generally varied from 12 to 80% while those for chlorinated pesticides varied from 0 to 60%. Estimates of method detection limit were based on 1) instrument detection limit, 2) sample injection volume, 3) final sample volume, 4) percent spike recovery, and 5) sample weight. With a GC–MS detection limit of about 200 pg/mL with 1 mL sample injection from a 1 mL sample concentrate and with approximately 30% analyte recovery, the method detection limit (approximately 10 g sample size) was estimated to be about 250 ppb. The analyte concentrations are expressed on a dry weight basis, and they are not corrected for recoveries of internal standards or spiked analytes. All positive values are reported and discussed in the results; however, those near the instrument or the method detection limit must be interpreted with proper caution.

There was considerable sample-to-sample variability in concentrations of organic contaminants in replicate sediments collected within a given station. This is evident in individual PAH concentrations in sediments collected from three replicate grabs at station NY6 in August 1987 (Fig. 2). Consequently, the temporal and spatial comparisons are based on mean concentration values for each station for each year. Also, to simplify

Table 1

GC/MS analyses of sediment extracts in select ion monitoring (SIM) mode.

Group (N)	Ions (N)	Start time	Compound (retention time)	SIM ions (% intensity)
1	4	26.0	naphthalene (31.053)	128(100), 51(13), 129(11), 64(11)
2	4	36.0	2-methylnaphthalene (36.684)	142(100), 141(85), 115(31), 143(11)
	4		1-methylnaphthalene (37.445)	142(100), 141(86), 115(34), 139(11)
3	4	39.0	biphenyl (40.666)	154(100), 153(31), 152(28), 76(17)
	4		2,6-dimethylnaphthalene (41.889)	156(100), 141(74), 155(34), 115(14)
	4		acenaphthylene (43.952)	152(100), 151(20), 76(17), 153(14)
	4		acenaphthene (45.439)	154(100), 153(86), 152(42), 76(23)
	3		PCB #1 (1 Cl) (45.981)	188(100), 190(33.4), 189(13.5)
4	4	47.5	2,3,5-trimethylnaphthalene (49.011)	170(100), 155(94), 153(23), 169(18)
	4		fluorene (50.161)	166(100), 165(80), 167(15), 163(12)
5	6	55.0	hexachlorobenzene (56.971)	284(100), 286(81), 282(55), 142(52), 288(37), 249(33)
	4		PCB #8 (2 Cl) (57.059)	222(100), 224(66), 223(14), 226(11)
6	4	59.5	phenanthrene (63.413)	178(100), 176(17), 179(16), 89(14)
	4		anthracene (64.293)	178(100), 176(17), 179(16), 89(14)
	7		lindane (61.379)	181(100), 183(98), 109(98), 51(91), 219(75), 111(100), 85(54)
	5		PCB #18 (3 Cl) (63.092)	256(100), 258(99), 260(33), 257(14), 259(13)
7	7	66.0	heptachlor (72.731)	100(100), 65(44), 272(40), 274(32), 102(34), 237(23), 270(15)
	5		PCB #29 (3 Cl) (68.043)	256(100), 258(99), 260(33), 257(14), 259(13)
	6		PCB #50 (4 Cl) (69.939)	292(100), 290(76), 294(49), 293(13), 291(10), 296(11)
	5		PCB #28 (3 Cl) (70.694)	256(100), 258(99), 260(33), 257(14), 259(13)
8	7	73.5	aldrin (78.591)	66(100), 79(43), 91(34), 263(32), 265(25), 101(40), 261(46)
	5		PCB #104 (5 Cl) (78.600)	326(100), 328(66), 324(61), 330(22), 327(14)
	5		PCB #44 (4 Cl) (79.608)	292(100), 290(76), 294(49), 293(13), 296(11)
	3		1-methylphenanthrene (75.096)	192(100), 191(56), 189(28)
	5		PCB #52 (4 Cl) (76.474)	292(100), 290(76), 294(49), 293(13), 296(11)
9	7	84.0	heptachlor epoxide (85.484)	81(100), 353(94), 355(72), 351 (48), 151(13), 263(13), 109(14)
	4		fluoranthene (86.499)	202(100), 203(19), 200(17), 101(14)
	6		PCB #66 (4 Cl) (87.133)	292(100), 290(76), 294(49), 293(13), 291(10), 296(11)
10	4	88.5	o,p'-DDE (90.708)	246(100), 318(79), 248(66), 316(61)
	5		alpha-chlordane (91.950)	373(100), 375(95), 371(50), 377(54), 237(58)
	5		t-nonachlor (92.531)	409(100), 407(85), 237(70), 411(48), 239(53)
	4		PCB #101 (5 Cl) (91.375)	326(100), 328(66), 324(61), 330(22)
	3		pyrene (90.949)	202(100), 203(26), 200(21)
11	4	94.0	p,p'-DDE (96.703)	246(100), 318(79), 248(66), 316(61)
	3		PCB #77 (4 Cl) (97.524)	292(100), 290(76), 294(49)
12	4	101.0	p,p'-DDD (105.227)	235(100), 237(65), 165(41), 236(17)
	5		o,p'-DDT (105.509)	235(100), 237(68), 165(38), 236(16), 176(38)
	5		PCB #118 (5 Cl) (102.930)	326(100), 328(66), 324(61), 330(22), 327(14)
	5		PCB #188 (7 Cl) (105.550)	394(100), 396(98), 398(54), 392(44), 400(18)
	5		PCB #153 (6 Cl) (107.849)	360(100), 362(82), 358(51), 364(36), 361(14)
	5		PCB #105 (5 Cl) (108.266)	326(100), 328(66), 324(61), 330(22), 327(14)
13	5	111.0	p,p'-DDT (112.694)	235(100), 237(68), 165(38), 236(16), 176(38)
	5		PCB #138 (6 Cl) (112.576)	360(100), 362(82), 358(51), 364(36), 361(14)
	5		PCB #126 (5 Cl) (114.180)	326(100), 328(66), 324(61), 330(22), 327(14)

continued

Table 1 (continued)

Group (N)	Ions (N)	Start time	Compound (retention time)	SIM ions (% intensity)
13	5		PCB #187 (7 Cl) (114.909)	394(100), 396(98), 398(54), 392(44), 400(18)
	5		PCB #128 (6 Cl) (116.320)	360(100), 362(82), 358(51), 364(36), 361(14)
14	4	117.2	benz(a)anthracene (118.766)	228(100), 229(20), 226(20), 43(18)
	4		chrysene (119.260)	228(100), 226(21), 229(20), 114(15)
	6		PCB #200 (8 Cl) (119.891)	430(100), 428(87), 432(66), 426(33), 434(27), 431(13)
	6		PCB #180 (7 Cl) (121.432)	394(100), 396(98), 398(54), 392(44), 400(18), 395(14)
15	6	122.6	PCB #170 (7 Cl) (124.198)	394(100), 396(98), 398(54), 392(44), 400(18), 395(14)
	7		PCB #195 (8 Cl) (128.022)	430(100), 428(87), 432(66), 426(33), 434(27), 429(12), 431(13)
	7		mirex (124.903)	272(100), 274(75), 270(54), 237(46), 239(47), 235(43), 276(32)
16	4	130.0	benzo(b)fluoranthene (131.661)	252(100), 253(23), 126(23), 250(19)
	4		benzo(k)fluoranthene (131.941)	252(100), 126(27), 253(23), 250(20)
	4		benzo(e)pyrene (134.054)	252(100), 250(24), 253(21), 125(16)
	4		benzo(a)pyrene (134.480)	252(100), 126(23), 253(21), 250(16)
	4		perylene (135.205)	252(100), 126(26), 253(21), 250(19)
17	4	141.0	indeno(1,2,3-cd)pyrene (142.870)	276(100), 274(25), 138(24), 277(22)
	4		dibenz(a,h)anthracene (143.172)	278(100), 279(24), 139(24), 276(16)
	4		benzo(ghi)perylene (144.667)	276(100), 138(37), 137(28), 277(25)

the graphical representation of over 60 individual compounds, the results are composited into three classes: PAH's, PCB's, and chlorinated pesticides (Table 2).

Results and Discussion

Concentrations of most organic contaminants in sediments from station NY6 were higher than those at station R2, which in turn were higher than concentrations at station NY11. Concentrations of PAH's also were generally higher than those of PCB's and chlorinated pesticides.

PAH's

Many potentially carcinogenic PAH's, including chrysene, benz(a)anthracene, benzofluoranthenes, benzopyrenes, perylene, indeno(1,2,3-c,d)pyrene, and dibenz(a,h)anthracene, were found in a number of samples, particularly in sediments collected from station NY6. These contaminants were present in sediments collected in pre-cessation (1986–87) as well as post-cessation periods (1988–89).

For each of the August cruises, the individual PAH concentrations were generally highest in sediments collected from station NY6, lowest in sediments collected from station NY11, and intermediate in sediments collected from station R2 (Fig. 3), confirming our expectations about this trend (based on historical information). Mean concentrations of PAH's were lower in 1989 compared with 1986 for all three study stations (Fig. 3). Total and individual PAH concentrations in sediments from station NY6 in August 1986, August 1987, and August 1988 were comparable to the historical data (Boehm, 1982); however, they were lower in 1989 compared with previous years (Fig. 4a, b).

PAH concentrations in August 1988 deviated from the expected temporal trend. Spikes or elevations in certain compounds were observed at all stations and were particularly distinct at station NY6 (Fig. 3). Similar spikes were observed in August 1988 in bacteria and trace metal levels at station NY6 (O'Reilly et al., 1995; Zdanowicz et al., 1995). These spikes could have been caused by 1) unauthorized dumping of contaminated sludge after December 1987, 2) unreported oil spills, 3) suspension and redistribution of existing contaminants, 4) estuarine discharge of contaminated suspended sediments, 5) spatial heterogeneity, or 6) sampling of more contaminated sub-surficial sediments instead of the surficial sediments (top 1 cm layer) previously collected.

PCB's and Chlorinated Pesticides

For each study year, the PCB (Fig. 5) and chlorinated pesticide (Fig. 6) concentrations were highest in sedi-

Table 2
Designations of various analytes in Figures 2–6.

Number	Polynuclear Aromatic Hydrocarbon
A11	Naphthalene
A12	2-Methylnaphthalene
A13	1-Methylnaphthalene
A14	Biphenyl
A15	2,6-Dimethylnaphthalene
A16	Acenaphthylene
A17	Acenaphthene
A18	2,3,5-Trimethylnaphthalene
A19	Fluorene
A20	Phenanthrene
A21	Anthracene
A22	1-Methylphenanthrene
A23	Fluoranthene
A24	Pyrene
A25	Benz(a)anthracene
A26	Chrysene
A27	Benzo(b)fluoranthene
A28	Benzo(k)fluoranthene
A29	Benzo(e)pyrene
A30	Benzo(a)pyrene
A31	Perylene
A32	Indeno(1,2,3-c,d)pyrene
A33	Dibenz(a,h)anthracene
A34	benzo(g,h,i)perylene
B35	PCB #1
B36	PCB #8
B37	PCB #18
B38	PCB #29
B39	PCB #50
B40	PCB #28
B41	PCB #52
B42	PCB #104
B43	PCB #44
B44	PCB #66
B45	PCB #101
B46	PCB #77
B47	PCB #118
B48	PCB #188
B49	PCB #153
B50	PCB #105
B51	PCB #138
B52	PCB #126
B53	PCB #187
B54	PCB #128
B55	PCB #200
B56	PCB #170
B57	PCB #195
B58	PCB #206
P59	Hexachlorobenzene
P60	Lindane
P61	Heptachlor
P62	Aldrin
P63	Heptachlorepoxide
P64	o,p'-DDE
P65	a-Chlordane
P66	trans-Nonachlor
P67	p,p'-DDE
P68	p,p'-DDD
P69	o,p'-DDT
P70	p,p'-DDT
P71	Mirex

ments collected from station NY6, lowest in sediments collected from station NY11, and intermediate in sediments collected from station R2.

Individual species of PCB's and chlorinated pesticides were not always present in every sample analyzed. Therefore, to assess temporal changes in sediment contamination, analyte concentrations were totaled by class, and the average concentrations of these classes of compounds were compared. For each station, the total PCB (Fig. 7) as well as total DDT-related pesticide (Fig. 8) concentrations were lower in 1989 than in 1986 and 1987. Similar to PAH results, a spike of total PCB's was observed in 1988 (Fig. 7); however, no such spike was observed for total DDT in 1988 (Fig 8).

Statistical Analyses

Representative compounds of the 24 PAH analytes are depicted for each station (Fig. 9) for the examination of 1) temporal trends, 2) mean individual contaminant concentrations, 3) variance in contaminant concentrations and 4) results of Duncan's multiple range test ($\alpha = 0.05$). These charts and the results of Duncan's multiple range test suggest (for these limited samples) that the contaminated stations (NY6 and R2) did not change significantly in the post-cessation period through August 1989 (Fig. 9). These analyses tentatively support our initial hypothesis that levels of PCB's (as a marker for organic contaminants in general) in sediment will remain essentially unchanged.

Comparison With Other Studies

In a related study, concentrations of trace metals in surface sediments from the highly contaminated station NY6 decreased significantly within one month of complete cessation of sludge dumping in December 1987; however, a buried reservoir of contaminants re-

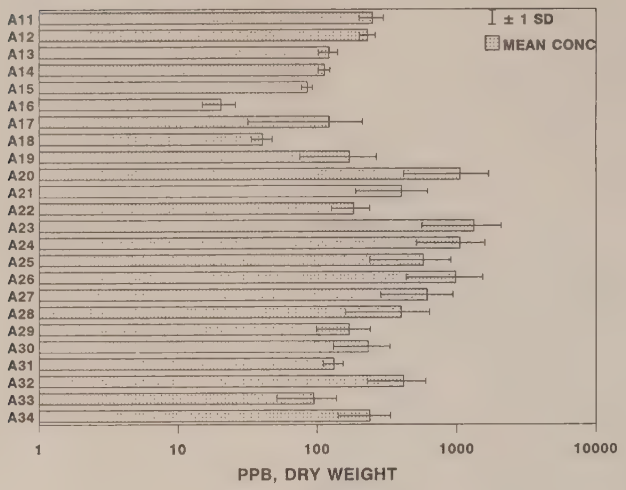

Figure 2

Mean SD of PAH (polynuclear aromatic hydrocarbon) concentrations in three replicate grabs at station NY6 in August 1987. Compound identifications are listed in Table 2.

Table 3

GC/MS operating parameters.

Initial values

Initial temperature	40°C
Equilibration time	10 min
Purge off time	1 min
Initial time	3 min
Run time	146 min

Temperature programming

Level	Heating rate (C min^{-1})	Final temp (C)	Final time (min)	Total time (min)
1	3	170	10.0	56.33
2	1	210	10.0	106.33
3	3	300	10.0	146.33

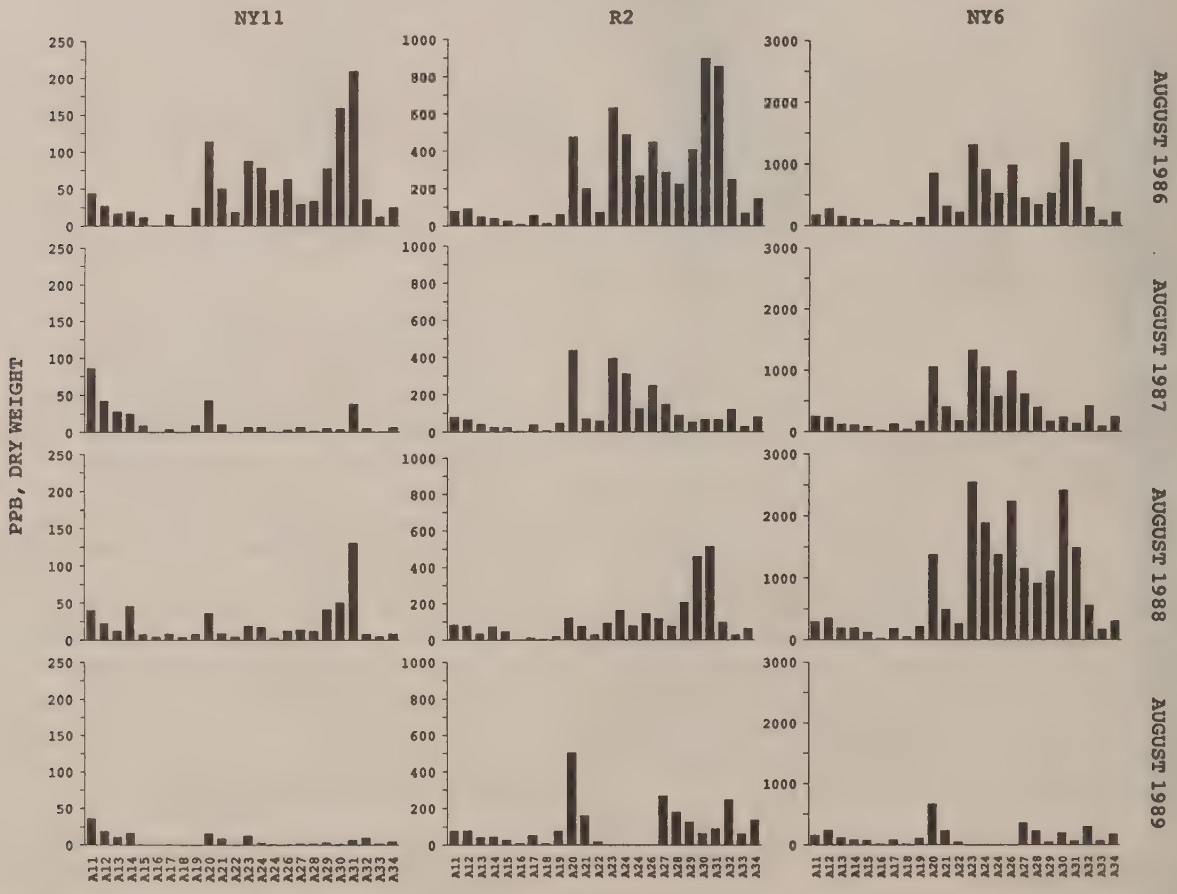

Figure 3

Mean PAH (polynuclear aromatic hydrocarbon) concentrations in New York Bight sediments (compound identifications are listed in Table 2).

mained at that site (Zdanowicz et al., 1995). Metal concentrations in surface and buried sediment from stations R2 and NY11 did not change significantly with cessation. The mean organic contaminant concentrations in surface sediments were lower in the post-cessation period than the pre-cessation period, although not significantly.

At the Philadelphia sewage sludge dumpsite, metal concentrations were reported to have declined rapidly after the cessation of dumping (Devine and Simpson, 1985). Similarly, subsequent to the cessation of dumping, metal concentrations in the surface sediments decreased at the sewage sludge dumpsite in Firth of Clyde, Scotland, but like the 12-mile dumpsite, metal concentrations in the buried sediments remained high (Clark et al., 1990). Organic analyses were not made in the Philadelphia or the Firth of Clyde studies.

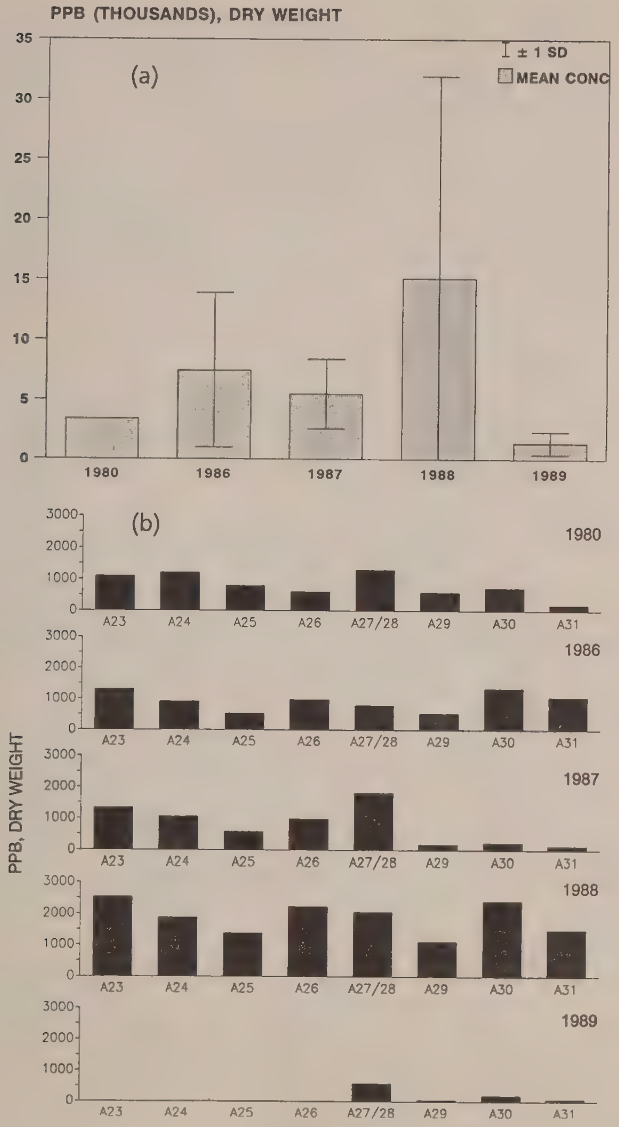

Figure 4

Time series for (a) total 4 and 5-membered PAH's (polynuclear aromatic hydrocarbons) at NY6 and (b) individual 4 and 5-membered PAH's at NY6 (1980 data from Boehm (1982); compound identifications are listed in Table 2).

Conclusions

Only a few of the available samples were analyzed for this paper relative to the larger New York Bight study area. In addition, the temporal scale of the study was relatively short. These factors combine to make detailed interpretations and conclusions difficult and risky. Nevertheless, it can be shown that:

1. There was considerable sample-to-sample variability in the organic contaminant concentrations at each of the three replicate stations;
2. In agreement with historical data, organic contaminants in sediments were higher in samples collected nearer the dumpsite (station NY6) than in those from two reference stations (R2 and NY11) farther from the dumpsite. For all samples, PAH levels were higher than PCB and chlorinated pesticide levels;
3. There was no apparent convergence of values between station NY6 and reference station NY11 by August 1989;
4. Spikes of PAH and PCB concentrations were observed at station NY6 in August 1988; and
5. Although mean organic contaminant concentrations in August 1989 were generally lower than those in August 1986, the difference was not statistically significant. Completion of remaining analyses may improve the statistical contrast between dumping and post-dumping periods.

Acknowledgments

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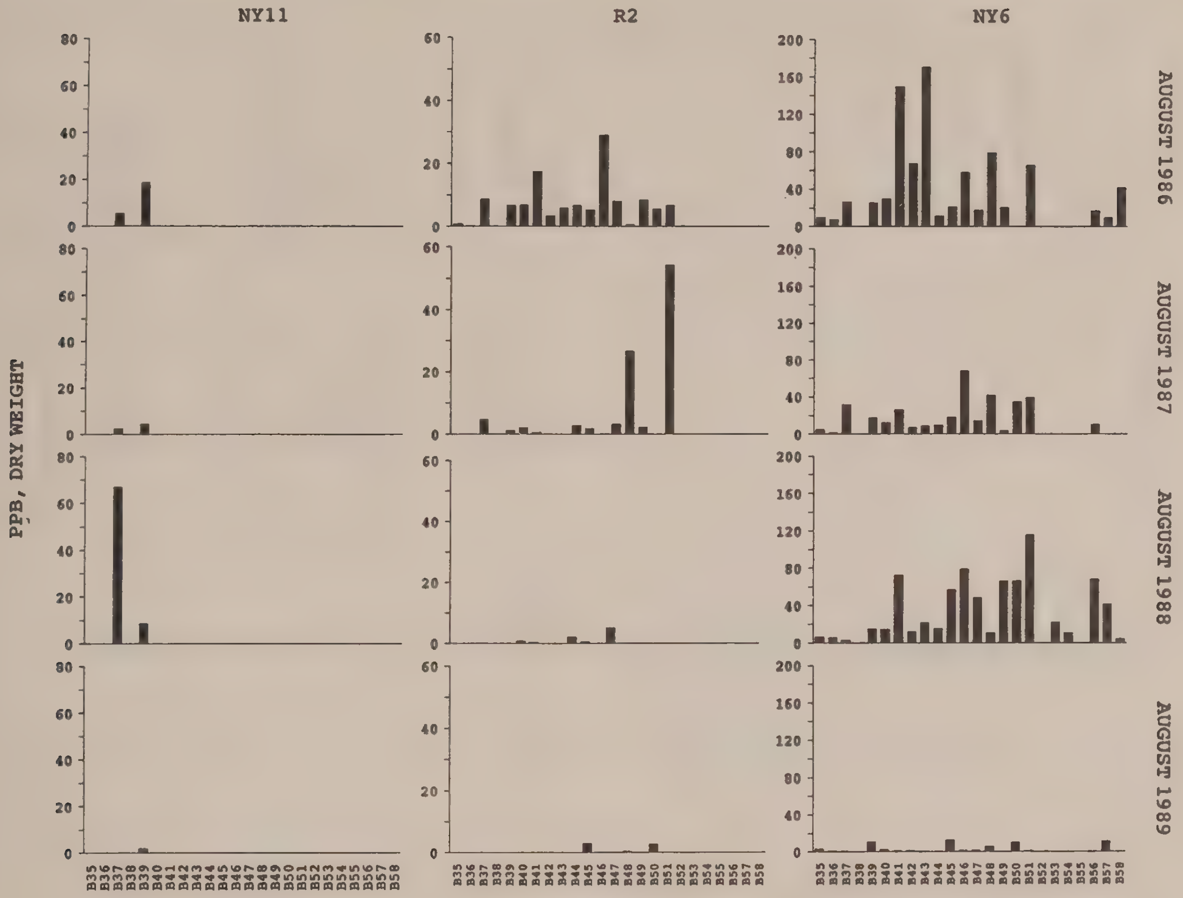

Figure 5
Mean PCB concentrations in New York Bight sediments (compound identifications are listed in Table 2).

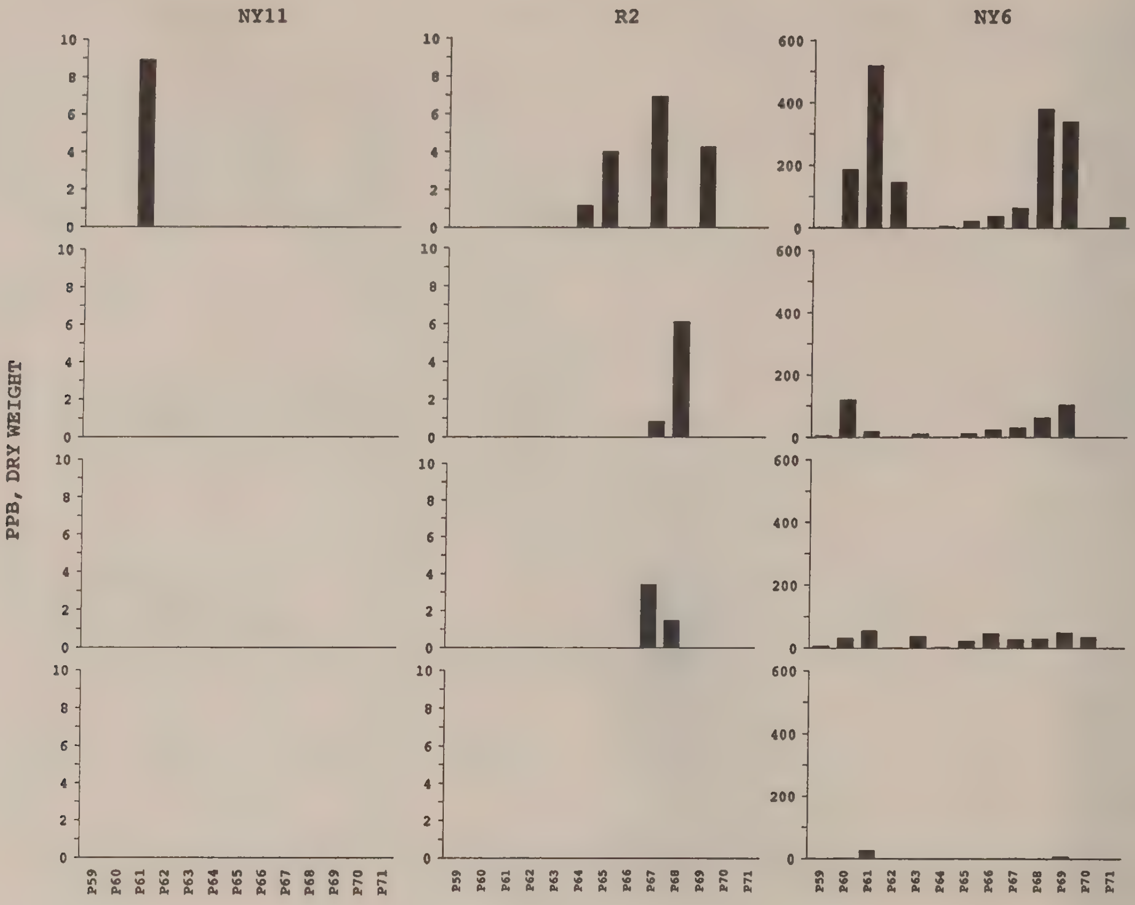

Figure 6
Mean chlorinated pesticide concentrations in New York Bight sediments (compound identifications are listed in Table 2).

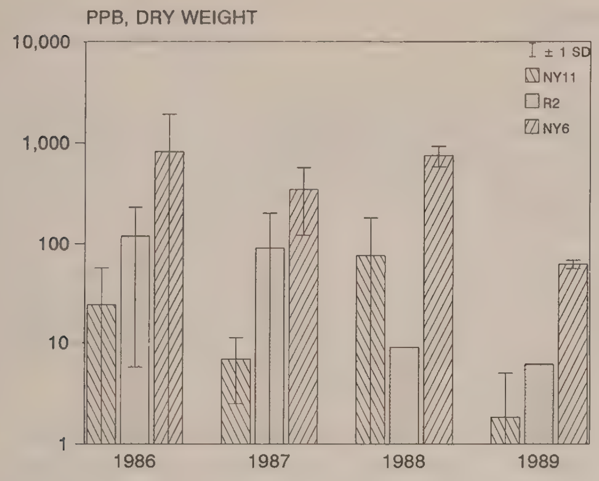

Figure 7
Time series for total PCB's.

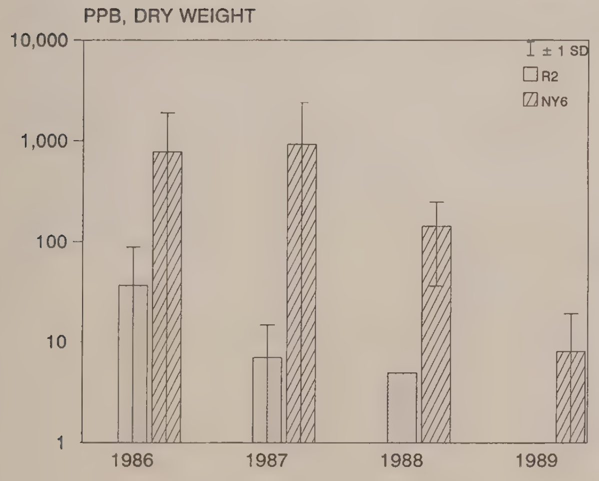

Figure 8
Time series for total DDT related pesticides.

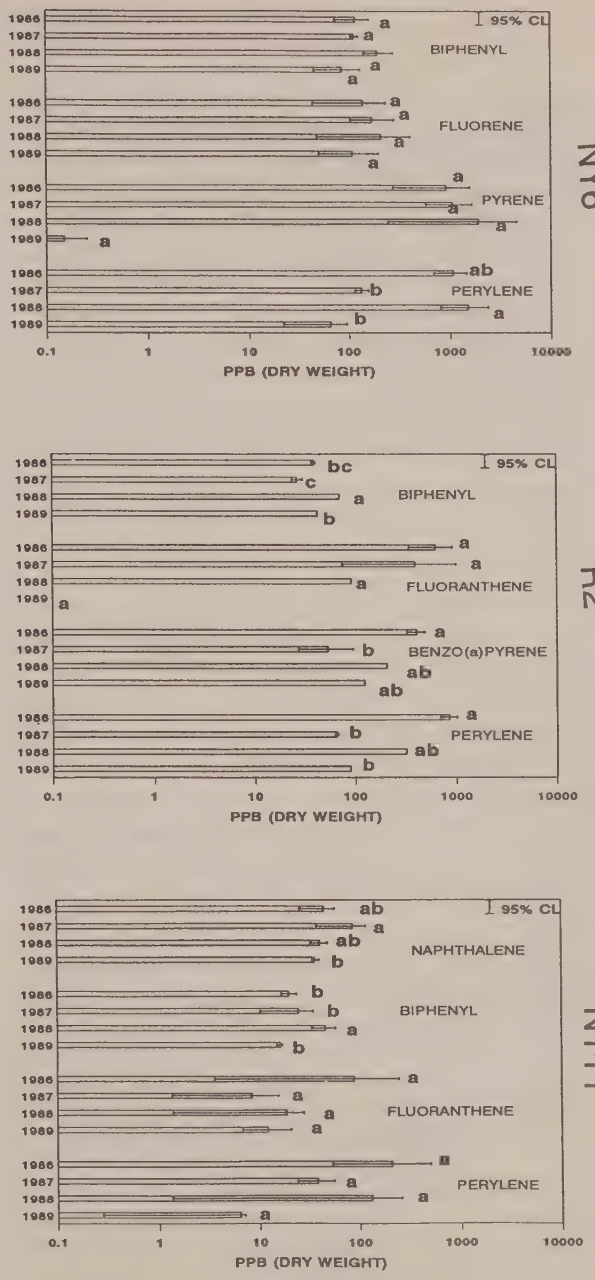

Figure 9
Time series for select PAH's (polynuclear aromatic hydrocarbons) in New York Bight sediments. For a given PAH analyte, within a set of four years, the mean concentrations designated by the same letter, such as: a, b, or c are not significantly different.

Audience Questions

Question: You showed decreases in the PCB's over time at the sludge dumpsite. I am following up on the point Jay O'Reilly made earlier. He showed there was a huge input of PCB's at the dredge disposal site due to dredge dumping, whereas at the sludge dumpsite PCB's are a very small proportion of the total. Do you have any data from the dredge spoil site in terms of PCB trends over time to relate to the cessation of the sludge dumping? Are the decreases that you see at the sludge site also associated with decreases at the dredge disposal site?

A. Deshpande: No, I do not have any data based on disposal of PCB's at the disposal site.

Question: If I understand you correctly, it looked to me like you were saying there were very low concentrations of DDT found at NY11. That would seem to be somewhat in contradiction with the discussion we had earlier that the primary source of DDT was from the atmosphere. If this is correct, I would expect the concentrations to be about the same at all the sites, assuming that the accumulative properties are about the same. If they are not, it would give me some concern that this was not a very good reference site.

A. Deshpande: Dick Bopp's work was on shelf break and most DDT's in that area were from aerosol-based material. I am not sure if he showed data on the ratio of DDT's to DDD's at NY11.

R. Bopp: The levels were low, only a few parts per billion, and you hit it right on the head. It is a net-depositional environment. It is a question of exactly how deep the recent depositional layer goes and how deep the bioturbation goes. But I saw no great difference. The lowest numbers are about a part per billion. Atmospheric inputs would give you two or three parts per billion.

A. Deshpande: The data were collected using gc/msd [gas chromatograph/mass selective detector], which is a little less sensitive than electron capture detection. The concentration might be just under detection limits and we might have missed it.

Question: You mentioned that you found essentially no significant differences over time, and yet I think you mentioned at the end that that may have been a function of sample size. Bob Pikanowski mentioned at the beginning that your sensitivity to finding differences is very much a function of the power of the test. I wonder if you looked at how large are differences you might expect to find given the samples that you have taken.

A. Deshpande: That is a very good question and we will answer it based on the data we are collecting right now. I did show these data to Bob Pikanowski, and he believed that statistically it did not make any difference; i.e., statistically the stations did not change, even though the average values showed that they were considerably lower in 1989 compared with those in 1986. Once we have more data points, the power of statistical analysis will be better exploited, and we will have a better understanding of the significance of cessation of dumping at the 12-mile site.

Changes in the Abundance and Distribution of *Clostridium perfringens*, a Microbial Indicator, Related to Cessation of Sewage Sludge Dumping in the New York Bight

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ABSTRACT

Concentrations of *Clostridium perfringens* spores, specific indicators of sewage contamination, were measured in surficial sediments from the 12-mile sewage sludge dumpsite and surrounding area to characterize responses to cessation of sludge dumping. Intensive, replicated sampling was conducted at three stations along a pollution gradient, at distances of 1.4, 4.1, and 11.1 km from the dumpsite. During dumping, *C. perfringens* spore concentrations were exceptionally elevated, generally 1.7×10^5 g^{-1}, at station NY6, the most polluted station nearest the dumpsite. These concentrations were 30 times greater than those observed at the most distant reference station, NY11, and six times greater than at our intermediate station, R2. Statistically significant decreases in spore concentrations, temporally related to the cessation of dumping, were observed at all three sites.

Decreases in spore abundance were also observed in surficial sediments collected over a large area surrounding the dumpsite. During dumping, approximately 27% of the 350 km^2 area surveyed had *C. perfringens* spore concentrations exceeding 1×10^4 g^{-1}; 3.3 yr following cessation, only 10% of the area surpassed these levels. The spatial distribution and temporal trends in *C. perfringens* abundance paralleled those observed for other sewage tracers.

Variation in the $\log_{10}$ of *C. perfringens* concentration was related directly to bottom depth and inversely related to distance from the dumpsite and time elapsed from the reduced dumping period. These associations were statistically highly significant and suggest that concentrations of *C. perfringens* in surficial sediments might decrease to New York Bight "background" levels 15–20 years after dumping ceased. While significant reductions in *C. perfringens* spore concentrations were observed in surficial sediments, subsurface concentrations, measured at depositional sites 3.3 yr after dumping ceased, were comparable to the elevated values observed in the surficial layer during dumping.

ABSTRACT (continued)

The responses of *C. perfringens* and coliforms to the effects of dumping are contrasted. Concentrations of coliform bacteria decreased more rapidly with distance from the dumpsite than did *C. perfringens.* Fecal coliform concentrations responded more rapidly to sludge dumping, decreasing during the reduced dumping period, and falling markedly after the dumpsite was closed. Concentrations of *C. perfringens* spores declined relatively slowly, persisting as a more conservative marker of residual sewage contamination.

Introduction

Coastal waters of the New York Bight (NY Bight) (Fig. 1) receive anthropogenic wastes generated by domestic and industrial activities of approximately 20 million people (Gross, 1976; Young et al., 1985). The three main sources of wastes are 1) outflow from the Hudson–Raritan estuary, 2) dumping of sewage sludge, and 3) disposal of dredged material from barges. Minor waste loadings emanate from the New Jersey and Long Island coastlines, and from aerial deposition (Stanford and Young, 1988). For some of these wastes, such as organic carbon, the Hudson–Raritan plume overwhelms the contribution from other sources (58% from the estuary, about 5% from sewage sludge; Mueller et al., 1976). For others, such as lead, dredge material is the largest source (51%), while (during dumping) sewage sludge (21%) and the estuary (24%) provided comparable inputs to the NY Bight (Stanford and Young, 1988).

The contribution to the NY Bight of microbial contaminants, such as fecal coliforms, from sewage sludge barged to the 12-mile dumpsite (Fig. 1) is insignificant (<0.01%) when compared with the contribution from unchlorinated municipal wastewater discharges and urban runoff from combined sewer overflows (>99%) (Mueller et al., 1976). This enters the Bight by the Hudson–Raritan estuarine plume which closely hugs the NJ coast. The plume is located over or seaward of the 12-mile dumpsite only twelve percent of the time (Fedosh and Munday, 1982). So, despite its large contribution of sewage contaminants to the NY Bight, the role of the estuarine plume in the contamination of sediments near the sewage sludge dumpsite is uncertain.

Dumping of sewage sludge began at the 12-mile site in 1924 (Gross, 1976; Gunnerson, 1988). Since then, at least 125×10^6 m^3 of sewage sludge was dumped, half of which occurred recently, from 1977 through 1987 (Fig. 2). The volume disposed of annually was quite small until 1958, when approximately 2.5×10^6 m^3/yr were barged to the site (Fig. 2). From 1958 through 1983, the volume tripled, reaching 7.5×10^6 m^3/yr, making it the world's largest oceanic sludge waste dumpsite (Devine et al., 1986). Beginning in 1981, the U.S. Environmental Protection Agency (USEPA) denied further applications for disposal of sewage sludge at the 12-mile site. In 1986, dumping decreased decrementally as the various permits for municipalities expired (Fig. 2, inset); on 31 December 1987, dumping ceased at the 12-mile site.

Several earlier studies, conducted by various agencies and institutions, identified significant sewage contamination of sediments and biological effects in the inner NY Bight (National Marine Fisheries Service, 1972; Pearce, 1972; Carmody et al., 1973; Gross et al., 1976; MESA, 1976; O'Connor et al., 1977; Mayer, 1982; Murchelano and Ziskowski, 1982; Reid et al., 1987). Based on elevated concentrations of microbiological indicators of sewage contamination, such as fecal coliforms, the area within an 11 km radius of the 12-mile site was closed in 1970 to the commercial harvesting of shellfish (Verber, 1976; Gaines and Reid, 1995). Considering the long history of dumping and the known accumulation of sewage-contaminated sediments adjacent to the dumpsite (Hatcher and McGillivary, 1979; Cabelli and Pedersen, 1982; Reid et al., 1982; Zdanowicz, 1991), decreases in the microbial contamination of sediments near the 12-mile site were expected following the cessation of sludge dumping (EPD, 1988). However, only a very rough guess was possible regarding the rate of decrease or the extent of areal improvement. The uncertainty was due to the abundance of microbial tracers of sewage (fecal coliforms and *Clostridium perfringens*) in the estuarine plume, in dredged harbor sediments dumped on the western margin of the Christiaensen Basin, and in sewage sludge. The exact role and influence of sludge dumping on habitat quality in the adjacent Christiaensen Basin, though believed predominant, could not be distinguished from these other potentially important sources (see "Introduction to Sediment Processes" in this volume; Wolfe et al., 1982). The closure of the dumpsite in 1987 provided the opportunity to resolve the role of sludge through time-series observations and resolve natural and seasonal variations from the anticipated responses related to the cessation of sludge dumping.

The purpose of this paper is to examine the temporal and spatial changes in *C. perfringens,* a highly specific tracer of sewage contamination, accompanying the cessation of dumping of sewage sludge in the NY Bight. *Clostridium perfringens* is a gram-positive, non-motile,

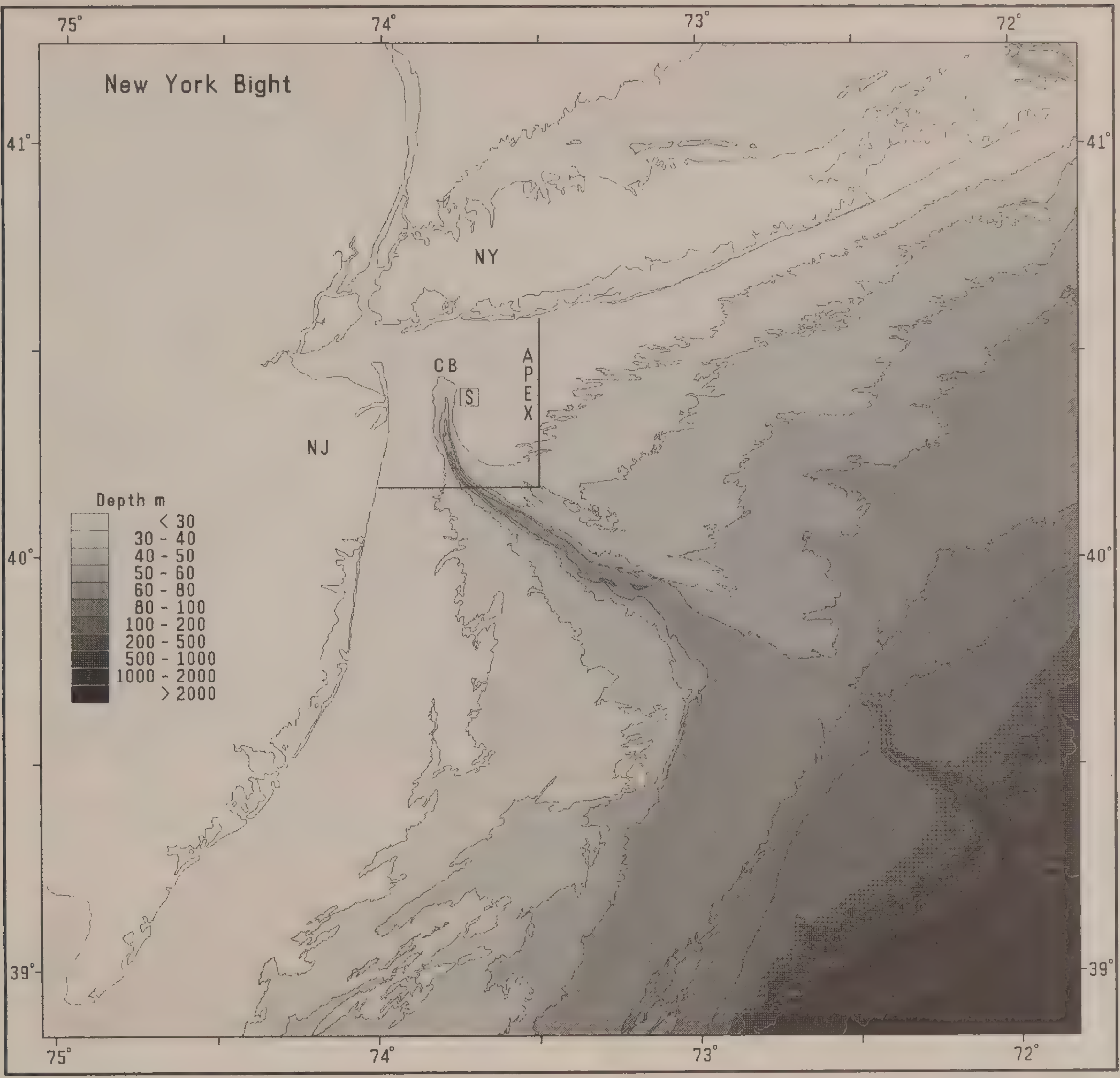

Figure 1

The New York Bight, with its major bottom topographic features and the locations of the Apex, the Christiaensen Basin (CB), and the 12-mile sewage sludge dumpsite (S).

rod-shaped bacterium, 0.9–1.3 μm by 3.0–9.0 μm in size. It is a strict obligate anaerobe found in fecal material. Under moderately adverse conditions it produces endospores that are less than 1.0 μm in diameter and can withstand extreme environmental conditions. For instance, *C. perfringens* spore counts drop less than one order of magnitude following chlorination typical of most sewage treatment facilities, whereas coliforms (e.g. *Escherichia coli*) and enterococci are reduced approximately four orders of magnitude (Miescier and Cabelli, 1982; DeBartolomeis, 1989). Previous field studies demonstrated that *C. perfringens* spores can be used as a conservative tracer for following the movement of sewage sludge (Bisson and Cabelli, 1979; Cabelli and Pedersen, 1982; Fujioka and Shizumura, 1983; Hill et al., 1993).

Methods

Sampling

This study was part of a multidisciplinary, multiagency program to determine the responses of the habitat and biota of the NY Bight to the cessation of sewage sludge

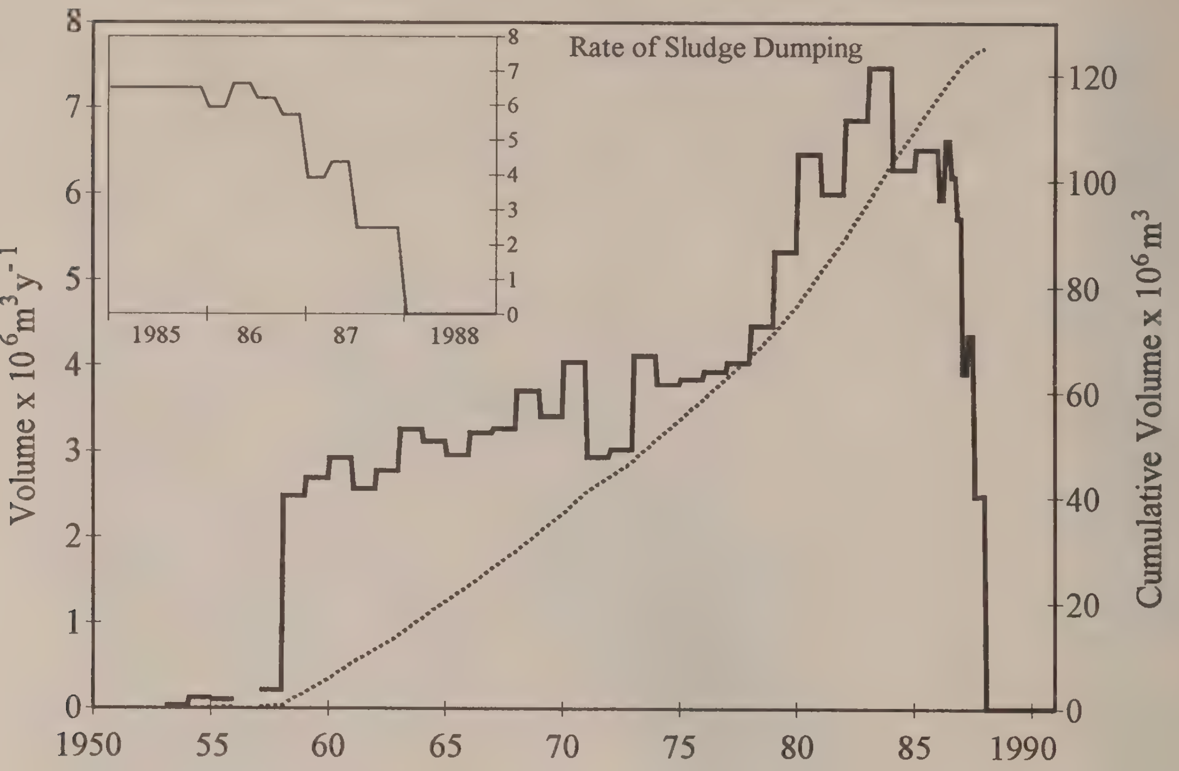

Figure 2

The rate of sewage sludge dumping at the 12-mile site between 1953 and 1991. Estimates for 1953–59 redrawn from Hatcher and McGillivary, 1979; estimates for 1960–72 from Mueller et al., 1976; data for 1973–87 provided by D. Pabst, US EPA, Water Management Division, Region II, New York, NY.

dumping at the 12-mile site (EPD, 1988). The study began formally in July 1986 while sludge dumping was decreasing (Fig. 2) and ended in September 1989, 21 months following the closure of the site. Sediment samples collected before and after this period are also included in our report. Two types of surveys, denoted replicate and broadscale, were conducted. During broadscale surveys, made every other month, sediments were collected at 25 stations (Fig. 3). Single grab samples of sediments were obtained at 22 stations and three grabs were obtained near the center of sites NY6, R2, and NY11 (Fig. 3). This scheme was employed to quantify changes in the distribution of sediment contaminants throughout a large area surrounding the dumpsite. *Clostridium perfringens* spore concentrations were measured in sediments collected during four of these broadscale surveys (April 1987, February 1988, April 1989, and March 1991). Replicate surveys, alternating with broadscale surveys, were conducted every other month, plus during August. During replicate surveys, eight grab samples were collected at each of three stations, NY6, R2, and NY11 (Fig. 3). *Clostridium perfringens* spore concentrations were measured in three of the replicate grabs which were taken at the center of each sampling site. The purpose of the replicate sampling scheme was to permit accurate estimates of temporal change and statistically test the null hypothesis of no change following cessation of sludge dumping (Pikanowski, 1995). The replicate stations were chosen to represent a pollution gradient within the Christiaensen Basin. They are in comparable water depths (26–30 m) but their distance from the dumpsite varies. Station NY6 is 1.4 km from the northwest corner of the dumpsite and represents the area where sewage sludge accumulated most heavily (Zdanowicz, 1991). The sediments are highly enriched in organic carbon and contaminants associated with fine-grained sediments. Station R2 is 4.1 km from the northwest corner of the sewage sludge dumpsite and has organically enriched and moderately contaminated sediments. Station NY11, at the southern end of the sampling area, is 11.1 km from the northwest corner of the sludge dumpsite. Sediments at NY11 were considered to be the least polluted at the start of the study (EPD, 1988).

Sediment samples were collected with a Smith-McIntyre grab (0.1 m^2) deployed at 20–25 m/min and

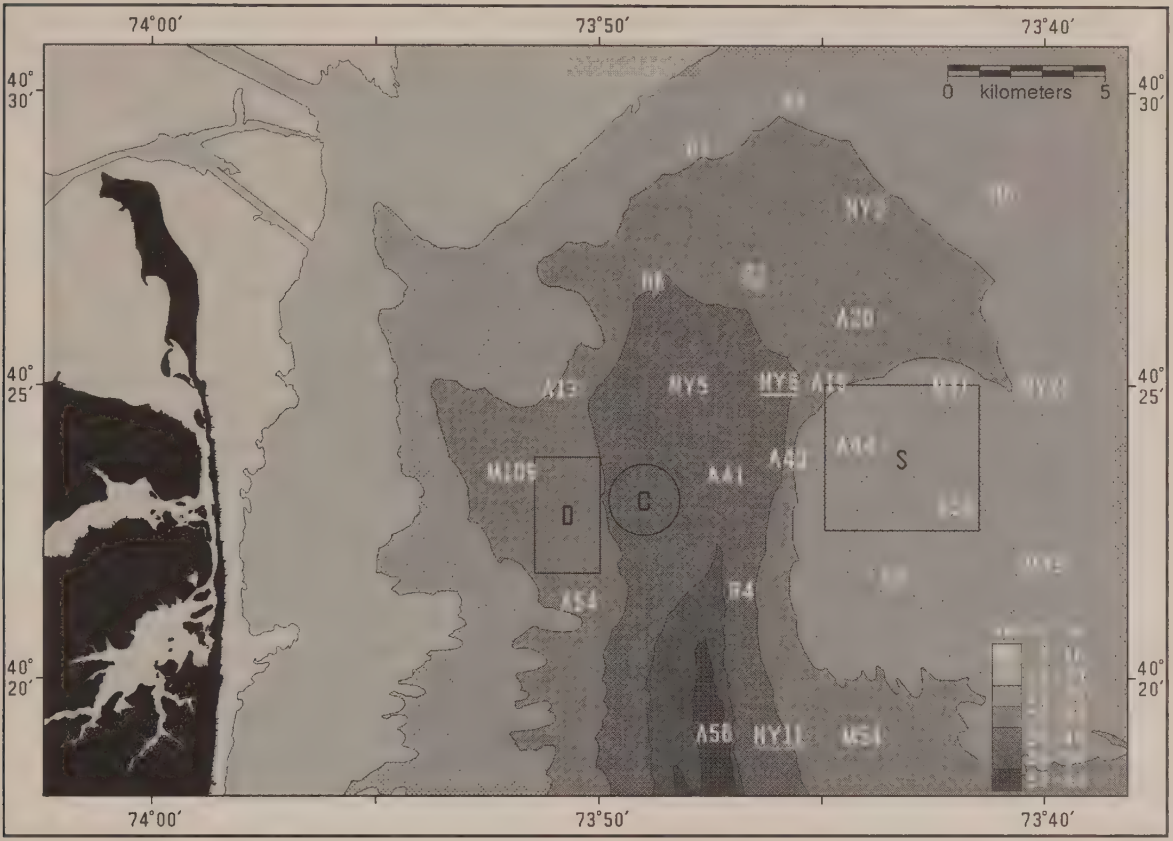

Figure 3

The locations of the 25 sampling sites in relation to bottom topography and the 12-mile sewage sludge (S), cellar dirt (C), and dredged material (D) dumpsites (The alphanumeric station codes are centered over the site coordinates). The "replicate" stations, NY6, R2, and NY11, are underlined. The Christiaensen Basin is delineated by the 25-m isobath.

retrieved at <10 m/min. If a coarse-grained sediment was expected, draining of interstitial and overlying water was prevented by setting the grab in a receptacle filled with water to provide back pressure. Disturbance of the surficial sediment and the overlying water by the ship's motion was minimized by placing a baffle in the grab immediately after it was on board. The baffle was prepared by breaking the bottom off of a sterile 125 ml plastic specimen cup (e.g. Elkay, Shrewsbury, MA) while it was still in its protective wrapper. This resulted in a sterile cylinder with an approximate diameter of 6 cm which was inserted approximately 2 cm (fractured side up) into the sediment and extended approximately 5 cm above the sediment surface.

After all sampling was completed that required the overlying water to remain in place, the jaws of the grab were opened slightly and the overlying water surrounding the baffle was allowed to drain away slowly. The baffle was then gently lifted and tilted slightly to permit the overlying water within the baffle to drain without disturbing the surface of the sediment. The baffle was removed and a sterile plastic scoop was used to sample sediment to a depth of 1 cm from within the ring formed by the baffle. Approximately 5 cm^3 of sediment were transferred to a sterile tube (NUNC, Naperville, IL), which was then stored at a temperature below 5°C until analyzed.

Vertical profiles of *C. perfringens* spore abundance were obtained from sediments collected in April 1991. A 3-cm diameter plastic core liner was inserted into the grab. The sediment cores were extruded and sectioned into 1-cm layers starting at the sediment-water interface and 1,3,5, and 7 cm below. Sections were placed in sterile cups and treated as above.

Laboratory Procedure

The extraction–separation procedure presented below follows the method outlined by Bisson and Cabelli (1979) and Emerson and Cabelli (1982). This is a membrane filter (MF) method which provides a direct count of the *C. perfringens* colonies that develop from spores on the surface of the MF in contact with mCP medium (Bisson and Cabelli, 1979). After thorough mixing, sediment samples were prepared by weighing out 1–3 g into 50-ml polycarbonate centrifuge tubes. Ten ml of sterile distilled water was added to the sample which was mixed using a vortex blender for 15 s. Particulates in the sample were disaggregated using a 500 W Tekmar Sonic Disrupter with a 3-mm tip set at 18 W. The duty cycle was set at 50% for intermittent operation and the sample was sonicated for 20 cycles or pulses. Care was taken and the controls were monitored to minimize cavitation which would have resulted in a shearing action, disrupting the *C. perfringens* spores and rendering them non-viable. Twenty-five ml of sterile distilled water was added to the centrifuge tube containing the sediment. This was mixed a second time for 15 s. The mixture was allowed to settle for 10 min whereupon decimal dilutions of the supernatant were prepared, using a phosphate-buffered solution (pH 7.2), to produce between 20 and 100 colonies per plate. The diluted samples were passed through a membrane filter (Millipore type HA, 0.45-µm pore size) which retained the spores. The MF were then placed on mCP growth media and incubated at 44.5°C for 24 h under anaerobic conditions (anaerobic jar BBL). The yellow *C. perfringens* colonies were counted using a stereo binocular microscope (Nikon SMZ-U) fitted with a fluorescent light source. Typical colonies were circular, 2–5 mm diameter, and slightly raised with a glossy surface. All plates were inverted over an open jar of ammonium hydroxide for a period of 30 s to confirm the presence of *C. perfringens* by a change in colony color from straw or yellow to pink or magenta. As a double confirmation, questionable colonies on filters were transferred to a plate of gelatin agar and incubated under anaerobic conditions for 1–2 h at 37°C. The filters were then removed from the gelatin medium which was flooded with Frazier's Reagent. Zones of hydrolysis (clearing) in the medium indicated the presence of gelatinase-positive *C. perfringens* colonies.

Sediment from the New York Bight collected in 1986 (sample NYB86) was used as a reference control, since samples were collected and analyzed over a span of several years. It was subjected to the same storage, extraction, separation, and counting procedures as the survey samples and consistently gave repeatable results.

Clostridium perfringens concentrations are reported as confirmed counts per gram (dry weight) of sediment (also Log_{10} count·g^{-1}). Spore counts per gram of sediment (wet weight) were converted to a dry weight basis using dry/wet conversion factors measured on parallel sediment samples which were weighed wet then oven-dried 24 h at 60°C and reweighed. Dry/wet conversion factors were not usually measured for the second grab from each of the three replicate stations (NY6, R2, NY11). Instead, for each triplicate set of grabs, the average dry/wet factor measured for the first and third grabs was used to correct spore counts measured in the second grab to a dry weight basis.

Results

Temporal Trends (Replicate Surveys)

Concentrations of *C. perfringens* spores in surficial sediments from the three replicate stations confirm the initial pollution-gradient hypothesis (EPD, 1988), which asserted that the degree of sewage contamination was greater at NY6 than at R2 and that R2 was more contaminated than the reference station NY11 (Fig. 4). Nearly all observations were made after mid–1986, when decreases in sludge dumping had begun (Fig. 2). Large variability among triplicate grab samples is evident during most surveys. The mean coefficient of variability of *C. perfringens* densities was 45%, 45%, and 66%, at NY6, R2, and NY11, respectively. The precision of the method, with no laboratory replicates, is expected to be about 25% (Bisson and Cabelli, 1979). The largest part of the within-station variability was probably due to field variability. This is reasonable in light of the relatively large variances observed over short distances in other geological, chemical, and biological properties (e.g. Zdanowicz et al., 1995). The seabed oxygen consumption data exemplify the variability that may occur over very small distances in the study area. The mean coefficient of variation in the rate of seabed oxygen consumption for 130 sets of cores taken simultaneously with a multiple corer, 1 m apart (Phoel et al., 1995), was 33% at station NY6 (Fromm[1]).

At station NY6, concentrations of *C. perfringens* spores were approximately 1.7×10^5 g^{-1} during the reduced dumping period, 1986–1987 (Fig. 4). By November 1991, nearly four years following cessation, spore concentrations had decreased one order of magnitude. Spore concentrations at NY6 were lower in 1986 than in 1987, deviating from the temporal pattern expected to accompany decreasing inputs of sewage sludge (Fig. 2). However, during the period of reduced dumping, changes occurred in the density and water content of sediments from NY6. The apparent increase from 1986 to 1987 may be an artifact resulting from these changes. During dumping, these sediments were frequently unconsolidated, black, and watery. At times they were oily muds having a consistency similar to mayonnaise (also noted by Hatcher and Keister, 1976).

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[1] S.A. Fromm, Northeast Fisheries Science Center, National Marine Fisheries Service, NOAA, Highlands, NJ. Pers. commun., January 1992.

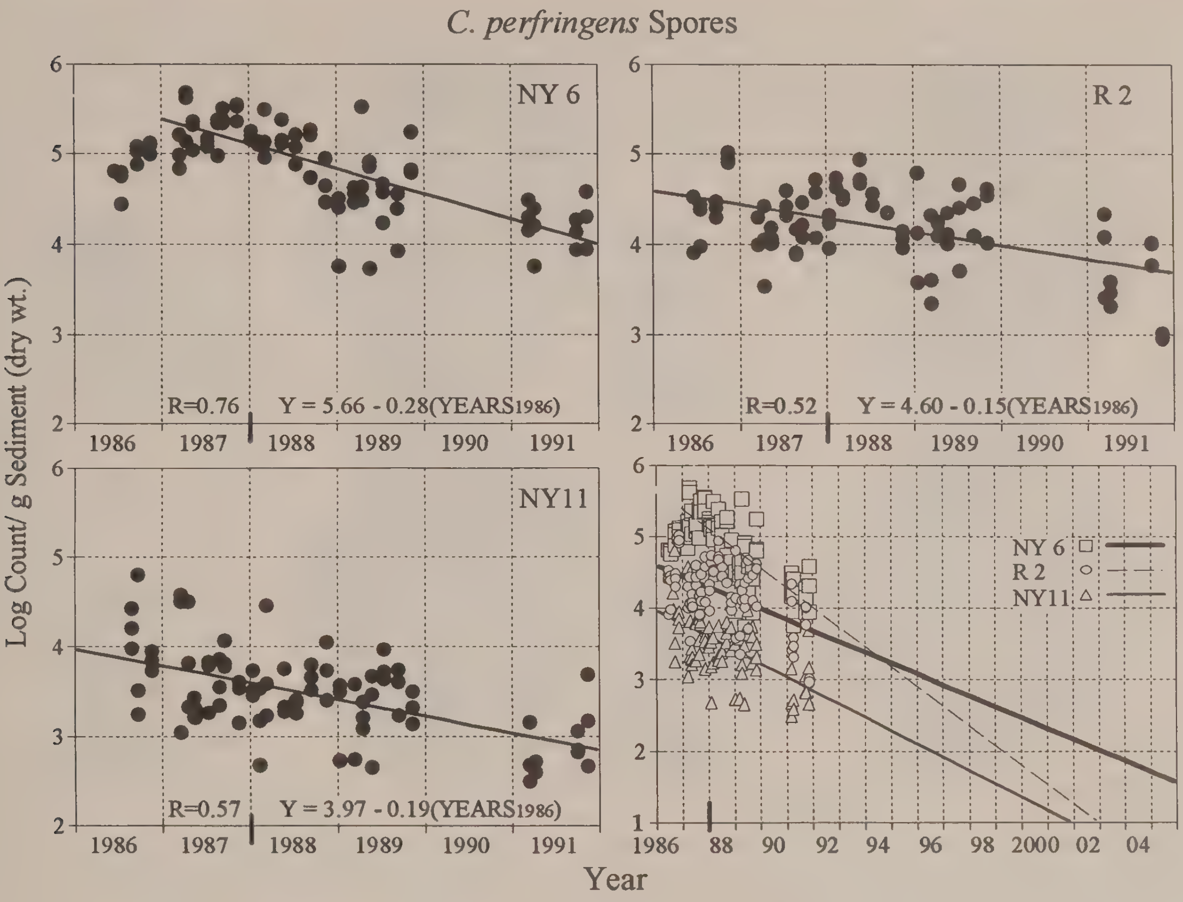

Figure 4

Changes in the concentrations of *Clostridium perfringens* in surficial sediment at replicate stations NY6, R2, and NY11 during the reduced dumping (1986–87) and post-dumping (1988–91) periods. The fitted line was obtained using the least squares *Y* regression model (the units of the *x*-axis are decimal years elapsed since 1 January 1986; *R*=correlation coefficient). Data in 1986 at station NY6 were not included in the regression (refer to text).

Concentrations of *C. perfringens* in surficial sediments from R2 were approximately 2.8×10^4 g^{-1} during the reduced dumping period (Fig. 4). By November 1991, spore concentrations had decreased to approximately 5×10^3 g^{-1}. There is a suggestion of an increase in spore counts at R2 during the first half of 1988. Possible reasons for this are discussed below. Concentrations of *C. perfringens* spores at station NY11 were approximately 6×10^3 g^{-1} during the reduced dumping period and 6.8×10^2 g^{-1} in November 1991 (Fig. 4).

The results of linear regression analyses of *C. perfringens* concentration versus time, from the reduced dumping period through November 1991, are given in Table 1. The temporal decreases in spores were highly significant at all three sites. The percent of variability ($r^2 \times 100$) explained by the linear regression model, for stations NY6, R2, and NY11, was 58%, 27%, and 32%, respectively. Station R2 had a lower temporal rate of decrease (slope = -0.15 y^{-1}) in spore concentration than that observed at NY6 (-0.28 y^{-1}) and NY11 (-0.19 y^{-1}). This analysis suggests that a 10-fold ($\log_{10}$) decrease in spores in surficial sediments would require from four to seven years.

Spatial Trends (Broadscale Surveys)

Clostridium perfringens spore concentrations in surficial sediments were generally high throughout most of the broadscale sampling area (Fig. 5). A similar pattern was evident from surveys during the reduced dumping period (1986–1987) and 1.1, 2.3, and 3.2 yr following closure of the site. The lowest spore concentrations (10^1–10^2) were observed in the northeast corner of the sampling area, at station R5, except during dumping, when values were higher (10^3–10^4). Relatively low val-

Table 1

Analysis of variance of the temporal decreases in *C. perfringens* at the replicate stations NY6, R2 and NY11.

Dependent Variable: Log$_{10}$ *C. perfringens* spores / g sediment (dry weight).
Independent Variable: Years$_{1986}$ (decimal years elapsed since 1 January 1986).

Analysis of Variance

Station	Source	DF	Mean Square	F Value	Prob >F	Dep. Mean	Correlation Squared
NY6	Model	1	10.524	**100.9**	**0.0001**	4.85	0.580
	Error	73[1]	0.104				
R2	Model	1	4.345	**30.6**	**0.0001**	4.20	0.267
	Error	84	0.142				
NY11	Model	1	6.358	**39.9**	**0.0001**	3.48	0.322
	Error	84	0.159				

Parameter Estimates of the Linear Regression Model[2]

Station	Variable	DF	Estimate	Std. Error	T for Ho	Prob > |T|
NY6	Intercept	1	5.659	0.0887	**63.8**	**0.0001**
	Slope	1	–0.275	0.0274	**–10.0**	**0.0001**
R2	Intercept	1	4.597	0.0827	**55.6**	**0.0001**
	Slope	1	–0.152	0.0274	**–5.53**	**0.0001**
NY11	Intercept	1	3.971	0.0895	**44.3**	**0.0001**
	Slope	1	–0.187	0.0296	**–6.32**	**0.0001**

[1] Data at station NY6 during 1986 were not included in the variance and regression analyses (refer to text).

[2] Statistics of the model were computed using the REG (Type II) procedure (SAS Institute Inc. 1988). (DF = degrees of freedom; T for Ho = Student's T test of the null hypothesis that the intercept or slope is zero; Prob> = probability of a larger value of |T|).

ues also were observed on the inshore and offshore flanks of the Christiaensen Basin, near the sewage sludge and dredge material dumpsites. Relatively high spore counts were found along the axis of the Christiaensen Basin at stations NY6, A41, and A56. During dumping, 65% of the 350 km^2 survey area had spore concentrations exceeding 10^3 g^{-1}, and 27% of the area exceeded 10^4 spores g^{-1}; whereas during the last survey, 55% and 10% of the area exceeded these respective levels (Fig. 6).

Vertical Profiles

In April 1991, cores from stations NY6, R2, NY11, A56, and NY22 were sectioned vertically to measure *C. perfringens* in the upper 10 cm of the sediment column (Fig. 7). The first four of these stations are considered highly depositional while sediments from station NY22, east of the Christiaensen Basin, consist of coarse sands. The relative differences in *C. perfringens* concentrations among the three replicate stations (i.e. the concentrations at NY6>R2>NY11) seen in the surficial sediment time series (Fig. 4) are also evident in the buried sediment layers (Fig. 7). At NY6 and R2, spore concentrations increased sharply, nearly 10-fold, from the 0.5 cm sample to the 1.5 cm sample. A more gradual, but comparable vertical increase in spore concentration was found at NY11. Low spore concentrations were observed at station NY22, only 7.4 km from the northwest corner of the sludge dumpsite. There is indication of an increase in spore concentration from 1.5 to 3.5 cm below surface. The coarse sandy sediment at NY22 limited the penetration of the grab to 4 cm, preventing a more complete depth profile.

Discussion

Comparisons with Other Studies

The highest concentrations of *C. perfringens* spores in sediments were found not within the sewage sludge dumpsite but in the adjacent Christiaensen Basin. This finding agrees with earlier reports by Cabelli and Pedersen (1982) and Graikoski (1982). The highest spore count of our study, 4.75 × 10^5 g^{-1}, was found at station NY6 during the reduced dumping period in April 1987. This surpasses levels measured in more enclosed, sewage-polluted coastal systems in the United States, such as Boston Harbor, MA (7.9 × 10^4); Salem

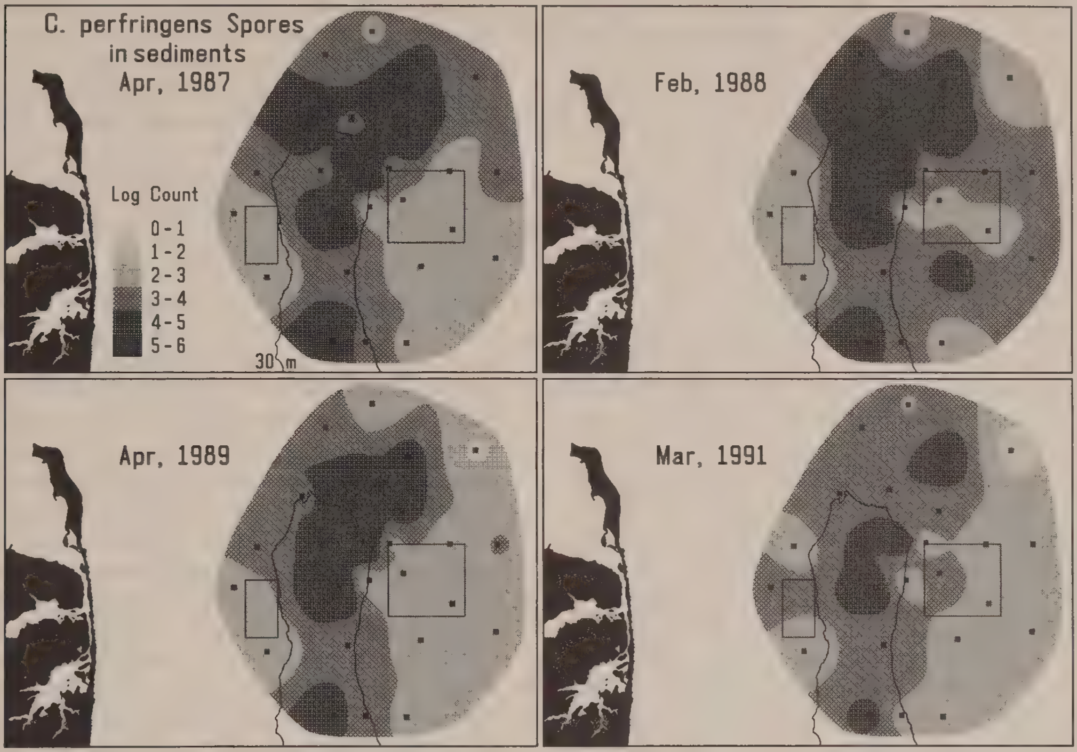

Figure 5

Contoured distributions of *Clostridium perfringens* spores in surficial sediments (Log$_{10}$ count/g sediment dry weight) during broadscale surveys in April 1987, February 1988, April 1989, and March 1991. The value contoured at stations NY6, R2, and NY11 (see Fig. 3) is the mean count from triplicate field samples; the results from single grab samples are used for the remaining sites. The boxes indicate the locations of the sewage sludge and dredge material dumpsites (see Fig. 3).

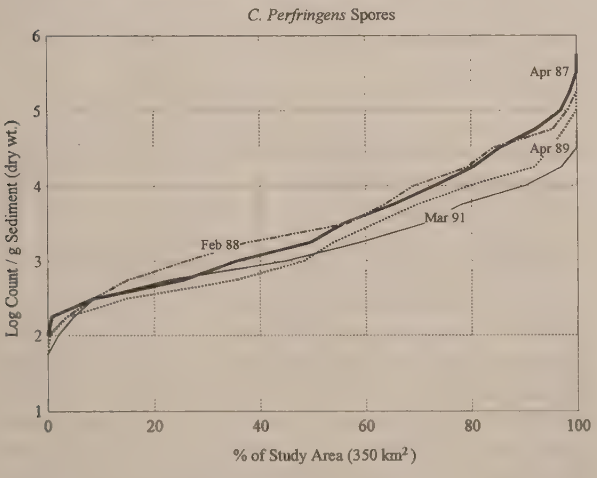

Figure 6

Cumulative plot of the percent of the study area versus *Clostridium perfringens* spore concentration. The computed areas were derived by integrating the contoured distributions of *C. perfringens* measured during broadscale surveys in April 1987, February 1988, April 1989, and March 1991.

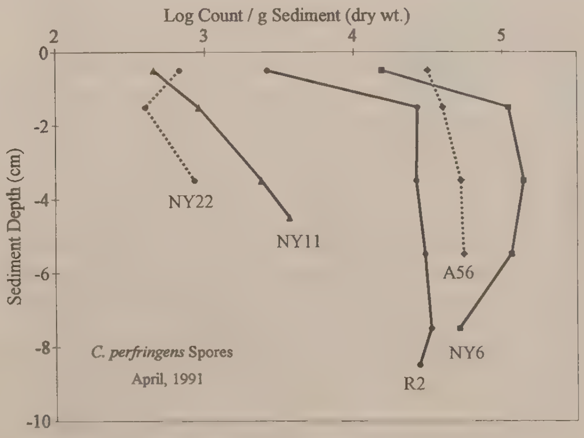

Figure 7

Vertical profiles of the concentration of *Clostridium perfringens* spores in sediments from selected stations sampled in April 1991, 3.3 years following the cessation of sludge dumping.

Harbor, MA (5.7×10^4); and Raritan Bay, NJ (2.4×10^4) (O'Connor[2]). The only sediments reported to exceed levels in the Christiaensen Basin are those from Black Rock Harbor, off New Haven Harbor, Conn., which had $3–6 \times 10^6$ g^{-1} of *C. perfringens* and a "strong asphalt and hydrogen sulfide smell." (Graikoski[3]).

During earlier studies of the NY Bight, Cabelli and Pedersen (1982) observed the highest *C. perfringens* concentrations ($1–2 \times 10^5$ g^{-1}) at a site near stations NY6 and A20 (Fig. 3). This compares very favorably with a number of values measured at station NY6 ($1–2 \times 10^5$ g^{-1}), during the reduced dumping period (Fig. 4). The lowest spore concentration (75 g^{-1}) was observed during the last broadscale survey, in the northeast corner of our sampling area (Fig. 5). This value approached the range (10–50 g^{-1}) found in offshore areas of the NY Bight, outside the Hudson Shelf Valley, and on Georges Bank (Cabelli et al., 1984).

Contamination from sludge dumped at the 12-mile site has not been limited to the Christiaensen Basin or the apex. *Clostridium perfringens* spore concentrations decrease progressively down the axis of the Hudson Shelf Valley towards the 200 m isobath, at the mouth of the Hudson Canyon (Cabelli and Pedersen, 1982; Graikoski, 1982; Davis et al., 1995). Even 100 km from the dumpsite, spore concentrations in the valley exceeded 100 g^{-1} sediment, or 10 times the "background" value. In part, this "far field" sewage contamination is the result of resuspension (Davis et al., 1995) of contaminated sediments from the Christiaensen Basin and subsequent down-valley transport during storm-induced flows (Manning, 1995). Other processes may augment the concentration of sewage contaminants along the axis of the Hudson Shelf Valley, such as the trapping of spores and fine particulate matter from the sewage sludge and estuarine plumes and from water flowing southwest over the valley (Beardsley et al., 1976; Ingham et al., 1982).

Earlier field studies of coastal sewage sludge dumpsites primarily used total and fecal coliforms as indices of sewage pollution. To our knowledge, *C. perfringens* has not been used previously to follow the detailed course of recovery of other sludge dumpsites. Direct comparisons with data from this study are not available, with the following exception. In 1985, five years after the Philadelphia sewage sludge dumpsite was closed, concentrations of 5–6 spores g^{-1} were measured in surficial sediments in the dumpsite by Duncanson et al. (1986). They attributed the exceptionally low values to cleaner sediments that were transported to the dumpsite from the mouth of the Delaware River.

Although different microbial tracers were used to follow recovery at the Philadelphia and 12-mile dumpsites, it is worth contrasting responses following the closure of these sites. Studies of the recovery of the Philadelphia sewage sludge dumpsite demonstrated that microbial indicators of sludge were present at the site during and after dumping, and tended to decrease with distance from the dumpsite (Devine and Simpson, 1985a). Studies conducted during the last year of dumping (1980) and during the following three years, revealed a pattern of higher total coliforms in sediments in the topographic low areas formed by the mesoscale ridges and swales (Devine and Simpson, 1985b). Based on findings of total and fecal coliform contamination in sediments (O'Malley et al., 1982), the U.S. Food and Drug Administration (FDA), in 1976, prohibited shellfishing in a 71 km^2 area centered over the dumpsite. In 1984, this area was reopened to shellfishing, following surveys showing detectable but low levels of coliform bacteria.

Important differences exist, however, between the Philadelphia site and the 12-mile site. The period of dumping sewage wastes at the Philadelphia site was much shorter (1973–1980), the volume dumped much less (4.11×10^6 m^3), the site was farther offshore (about 50 km), and over deeper water (about 50 m) compared with the 12-mile site (64 yr; 125×10^6 m^3; 20 km; 22 m). Furthermore, the Philadelphia site does not have a nearby deep depression such as the Christiaensen Basin. Consequently, no evidence of significant sludge accumulation or very degraded sediments was found at the Philadelphia site (Devine and Simpson, 1985b). This is in sharp contrast with our findings for the 12-mile dumpsite.

Temporal Trends Related to Sewage Sludge Dumping

Decreases in the concentrations of *C. perfringens* in surficial sediments at all three replicate stations were significantly correlated with time elapsed since the reduced dumping period (Table 1), and paralleled the decreasing trends in metals (Zdanowicz et al., 1995), organic contaminants (Deshpande and Powell, 1995) and increasing redox potential (Draxler, 1995). These temporal trends and the extensive areal changes observed at broadscale stations are largely the consequence of cessation of sludge dumping. Other possible causes for these changes seem improbable. Spore inactivation *in situ* or during sample storage, leading to germination failure during the laboratory assay, seems an unlikely explanation for the observed temporal decreases at the replicate stations. Studies of the effect of storage of sediments at 4°C and below freezing revealed insignificant changes in spore viability over 48 months following collection (Duncanson et al., 1986). Because *C. perfringens* spores are reported to survive well in sediments and are viable up to at least 300 yr (Granberg, 1983), it seems also unlikely that a large fractional reduction in viability of spores would occur during the few years following abatement of sludge dumping. However, the relationship between germination efficiency and age of the spores is not known. Bottom temperatures >20°C could force germination of spores and subsequent mortality (or grazing losses) of the resulting vegetative cells (assuming that these vegetative cells in turn do not sporulate). The 1986–1989 monthly record of bottom temperature shows peak temperatures reaching 17°C in September. Bottom temperature measured by current meters deployed at sites in the basin near station NY6 briefly approached 19°C during five days in September 1989 (Mountain and Arlen, 1995). Because bottom temperatures did not exceed 20°C in the deeper portions of the Christiaensen Basin, temporal changes in *C. perfringens* at the replicate stations do not appear to be confounded by temperature effects. However, this could potentially be a contributing factor in the diminution of spore counts during the post-dumping period in the shallow areas flanking the Christiaensen Basin which experience temperatures above 20°C (Mountain and Arlen, 1995). Loss of spores from grazing by microfauna is also possible; however, to our knowledge, no information exists concerning *in situ* grazing on *C. perfringens* spores.

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[2] O'Connor, T. National Status and Trends Benthic Surveillance Project, National Ocean Service, NOAA, Rockville, MD. Pers. commun., March 1992.

[3] Graikoski, J. T. 1984. Distribution of *C. perfringens* and *Vibrio* spp. in the northwest Atlantic, 1983. Annual progress report, Northeast Monitoring Program (NEMP). Northeast Fisheries Center, Woods Hole, MA.

Factors Influencing the Distribution of *C. perfringens*

Devine et al. (1986) compared the rates of loading and the local and regional tendencies of eight coastal sludge dumpsites (NY Bight, Thames, Firth of Clyde, Liverpool Bay, German Bight, Bristol Channel, Lyme Bay, and Plymouth) to accumulate sludge particles. Though the 12-mile site received the largest loadings of sludge solids, as much as 80% of the sludge dumped annually at the 12-mile site is dispersed away from the site (Devine et al., 1986). Despite this high rate of dispersion, the NY Bight has the highest estimated index of regional accumulation of sludge contaminants owing to the proximity of sludge dumping to the depression formed by the Christiaensen Basin and the Hudson Shelf Valley. Based on the distribution of the ratio of carbohydrate/total organic carbon in sediments, Hatcher and Keister (1976) stated: "[that] sewage is being dispersed throughout the apex and that it accumulates mostly in topographic lows." In these areas, the sediments are black silt, or black silty sands with elevated organic carbon between 1% and 4% (Hatcher and McGillivary, 1979). Thus, bottom topography in the NY Bight plays an important role in the deposition and retention of sewage-derived contaminants.

Cabelli and Pedersen (1982) observed that *C. perfringens* concentrations increased with bottom depth in the area north of our station A20 (Fig. 3), between the northern portion of the Christiaensen Basin and the Long Island coast. The rate of increase was approximately 0.23 $\log_{10}$ *C. perfringens* m^{-1}. A similar spore–bathymetric association is evident in our data, which partially overlap the area surveyed by Cabelli and Pederson, and include sediments east and west of the Christiaensen Basin and along its major axis. Spore concentration was correlated directly with increasing bottom depth. The functional linear regression slopes (Ricker, 1973; Laws and Archie, 1981) of the $\log_{10}$ of spore concentration versus depth are 0.25, 0.23, 0.21, and 0.18 m^{-1}, respectively, for broadscale surveys in 1987, 1988, 1989, and 1991 (Fig. 8). The slopes computed using the "least squares *y*" regression method were systematically lower (Fig. 8; see LSxy and LSy). The decrease in the regression slope over the four years results from disproportionately larger decreases in spore concentrations at the most contaminated sites than at the least contaminated sites.

Obviously, bottom depth does not explain all spatial variation in spore concentrations, otherwise stations NY6, R2, and NY11, which are between 26 and 30 m, would have comparable levels of spores, even though they are 1.4, 4.1, and 11.1 km from the northwest corner of the dumpsite. Indeed, Cabelli and Pedersen (1982) observed that *C. perfringens* concentrations decreased with increasing distance from the sewage sludge dumpsite. Sewage sludge particulates are very fine, constitute about 2–8% of the sludge by weight (Santoro and Fikslin, 1987) and are diluted rapidly, several thousandfold, shortly after dumping (Calloway et al., 1976; Proni et al., 1976). In situ observations by divers indicate that the coarser fraction of the sludge may reach the bottom rapidly near the dumpsite (Phoel[4]). In general, however, the slow settling rate of the fine particulate matter results in further dilution as it is transported away from the dumpsite by ambient currents (Draxler, 1979; Devine et al., 1986).

We explored statistically this distance–spore relationship using the broadscale survey data shown in Figure 5. Temporal changes evident in this series and in the replicate time-series (Fig. 4) were also considered. Accordingly, the statistical model assumes that variations in the concentration of *C. perfringens* should be related to bottom depth, distance from the sludge dumpsite, and time elapsed since the reduced dumping period. Analysis of variance of this model indicates that the independent variables, bottom depth, and distance from

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[4] Phoel, W. National Marine Fisheries Service, NOAA, Silver Spring, MD. Pers. commun., June 1991.

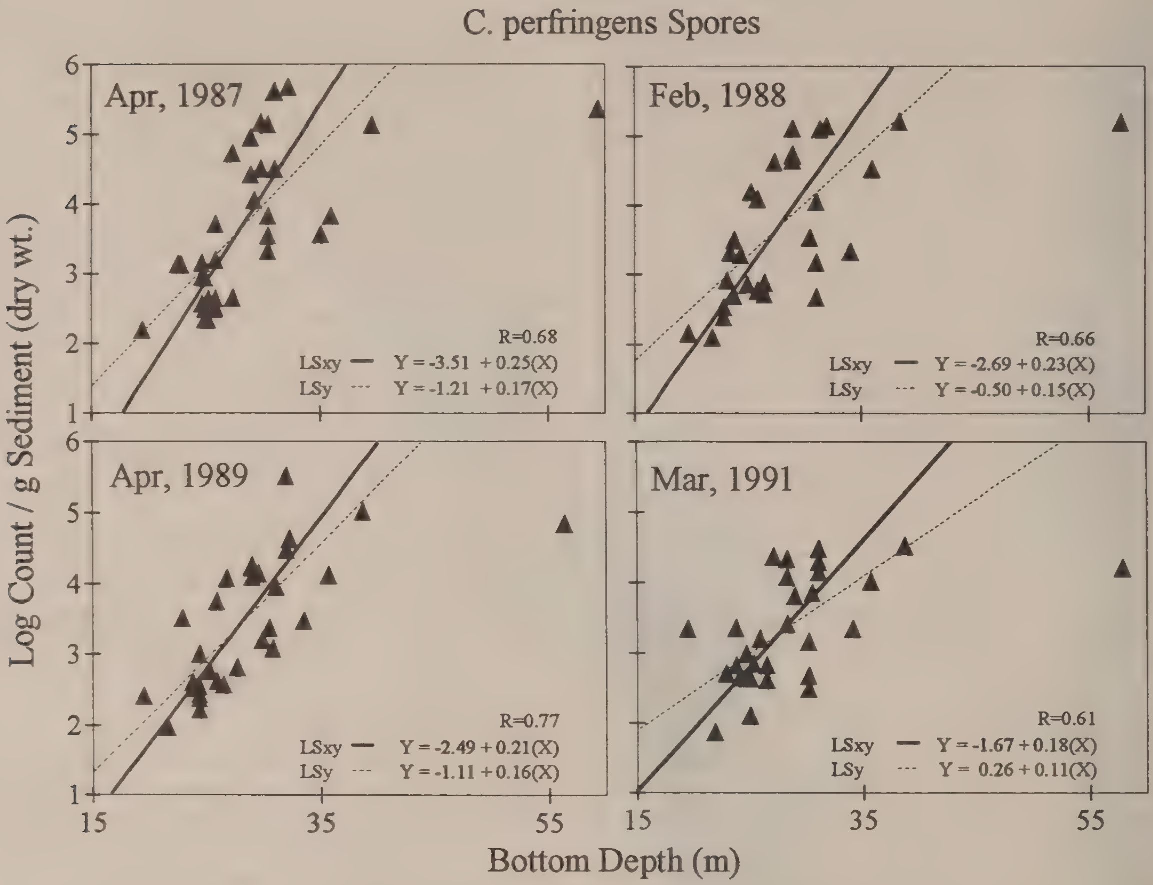

Figure 8

The relationship between *Clostridium perfringens* spore abundance in surficial sediments versus bottom depth during broadscale surveys in April 1987, February 1988, April 1989, and March 1991. (Observations at ~57 m, station A56, were excluded from the regression analyses; refer to text). LSy: least squares *Y* regression; LSxy: functional regression.

the northwest corner of the sludge dumpsite collectively account for 59% of the total variance (Table 2). The third independent variable, years elapsed since 1986, explains an additional 4% of the total variance. The results from this model are highly significant ($P = 0.0001$). The estimates of the regression slopes for each of the independent variables in the model are also highly significant. Therefore, over the entire study area, *C. perfringens* spore concentrations increase 10-fold with an 7.8 m increase in bottom depth, decrease 10-fold over a distance of 11 km from the northwest corner of the sludge dumpsite, and will decrease 10-fold over the 11 yr following cessation.

Other Sources of Variability

The vertical gradients in *C. perfringens* spores found at depositional stations in the Christiaensen Basin, 3.3 yr following the closure of the dumpsite, suggest that spore concentrations (and other sewage contaminants) in surficial sediments may increase temporarily in these areas following major storms which erode the surficial layer. The thickness of the sludge-contaminated layer was not routinely gauged. However, based on a single survey in April 1991, a layer of highly contaminated sediments, at least 8–9 cm thick, was present at several sites in the Christiaensen Basin. The sludge-contaminated layer is probably much thicker in some areas of the basin. Measurements of coprostanol, a fecal sterol, in sediment cores taken near NY6 during the mid–1970's demonstrated a highly sewage-enriched stratum extending from surface to a depth of 36–39 cm (Hatcher and McGillivary, 1979). The sustained elevated levels of *C. perfringens* found a few centimeters below the sediment surface in the Christiaensen Basin are concordant with the data of Reid et al. (1991) who reported that the concentration of tomato seeds (a marker of

Table 2

Model of *C. perfringens* concentrations in sediment as a linear function of bottom depth, distance from the northwest corner of the sewage sludge dumpsite, and time elapsed since 1986.

Dependent Variable[1]:	$\log_{10}$ *C. perfringens* spores / g sediment (dry weight).
Independent Variables:	Bottom Depth (m); Distance from NW corner of the sewage sludge dumpsite (km); Years_{1986} (decimal years elapsed since 1 January 1986).

Analysis of Variance

Source	DF	Mean Square	F Value	Prob>F	Dep. Mean	Correlation Squared
Model	3	20.192	**50.33**	**0.0001**	3.491	0.559
Error	119	0.401				

Parameter Estimates of the Linear Regression Model[2]

Variable	Estimate	Std. Error	T for Ho	Prob>|T|	Partial[3] Correlation
Intercept	0.714	0.457	1.561	0.1211	
Bottom Depth(m)	0.129	0.014	9.127	**0.0001**	0.412
Distance (km)	–0.090	0.017	–5.061	**0.0001**	0.177
Years_{1986}	–0.091	0.039	–2.330	**0.0215**	0.044

[1] Station A56 was excluded from this analyses (refer to text).

[2] Statistics of the model were computed using the REG (Type II) procedure (SAS Institute Inc. 1988). (DF = degrees of freedom; T for Ho = Student's T test of the null hypothesis that the estimated parameter is zero; Prob> = probability of a larger value of |T|).

[3] Partial correlations (Type II model) are squared. The covariance of estimates were essentially zero for all combinations of the independent variables.

sewage sludge contamination) in grab samples remained unchanged at NY6 following the cessation of sludge dumping. (Since the entire Smith-McIntyre grab sample is sieved, and depths of 10–12 cm are typically sampled, any changes in the tomato seed count in the surface veneer will be masked in the total seed count for the grab.) Trace metal concentrations also persisted at high levels in subsurface (5 cm) sediments while metal concentrations in the surficial layer at NY6 decreased significantly (Zdanowicz et al., 1995). Therefore, high concentrations of spores, comparable to the elevated levels recorded during dumping, may be observed episodically until the highly contaminated stratum is totally eroded or permanently capped with cleaner sediments.

An example of an elevated spore count event due to an erosion episode may have happened in early 1988. A major storm occurred 12–16 February 1988, during which wind speed reached about 40 knots (20.4 m/s). Wind speed was sustained at 33 kt (17 m/s), from the northeast, over 26 h beginning 13 February. Current meters were not operating during this event (Manning, 1995). However, Manning[5] suggests that this was potentially one of the more significant sediment resuspension events during the study, surpassing the May 1987 storm. If the surficial layer in depositional areas was resuspended by the February storm and the more contaminated substratum was exposed, then perhaps the increases in spore concentrations measured in early 1988 at R2 (Fig. 4) are explained by this mechanism. However, similar increases in spores, expected at NY6 and NY11, were not obvious (Fig. 4).

Major storms, such as the February 1988 event, may also generate temporal variability in spore concentrations in surficial sediments flanking the basin. This would result from the resuspension of contaminated sediments in the basin and their subsequent deposition onto the less-polluted surrounding sea floor. Prior to the February 1988 storm, spore concentrations at eight of the 11 sites sampled (A6, NY3, R1, R5, A19, A20, NY6, A56, M54, NY11, and R2) were lower than those measured during the 1987 survey, whereas 8 of the 14 remaining stations sampled after the storm (A44, A50, M31, M49, NY22, R6, A13, A41, A43, A54, M109, NY5, R3, and R4) had higher counts than in 1987. The greatest relative increases were on the seaward (east) side of the Christiaensen Basin (Fig. 5). The overall effect of this storm is reflected as an increase in the cumulative area having intermediate (about 1×10^3) *C. perfringens* concentrations (Fig. 6).

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[5] Manning, J. Northeast Fisheries Science Center, National Marine Fisheries Service, NOAA, Woods Hole, MA. Pers. commun., March 1992.

Spatial differences in the rate of sedimentation may explain some of the variability observed during our study. Based on vertical profiles of coprostanol and the observation that total organic carbon in sediment cores decreased with increasing depth and with the reduced volumes of sludge dumped in the 1950's and 1960's (Fig. 2), Hatcher and McGillivary (1979) concluded that sewage contamination dominated sedimentation in the Christiaensen Basin during the past several decades. Bopp et al. (1995) measured the net rate of fine-particle accumulation in the Christiaensen Basin (stations NY5, NY6, A41, A56; Fig. 3) and at adjacent sites in the upper Hudson Shelf Valley. Based on their measurements of the depth to which ^{137}Cs was found, these areas accumulated between 12 and 21 cm of sediment since the early 1950's, except A56, which accumulated between 40 and 200 cm in the several cores taken. This exceptionally high rate of sedimentation may explain the lower-than-expected *C. perfringens* concentrations observed consistently at A56 (Fig. 8). In addition to presumably receiving lower amounts of sewage-derived particulate matter than stations closer to the sludge dumpsite (e.g. NY6), sewage particulate matter reaching A56 was probably diluted with large amounts of non-sewage particles during deposition. Bopp et al. (1995) suggest the dredge material dumpsite as the likely source of these sediments.

Microbial Tracers of Sewage: Coliforms versus *C. perfringens*

Although not used routinely as a criterion for public health and shellfish quality, *C. perfringens* levels are correlated positively with fecal coliform concentrations in sediments (Watkins and Cabelli, 1985). The distribution of coliforms in sediments in the vicinity of the 12-mile site during a 1971 survey by FDA (Verber, 1976) is similar, at a coarse scale, to the general pattern observed for *C. perfringens* during the four broadscale surveys. Coliform concentrations were highest (1 × 10^3 Most Probable Number [MPN] g^{-1}) around the dumpsite and immediately west, in the Christiaensen Basin, and decreased three orders of magnitude 11 km away from the dumpsite. The most obvious difference is that coliform concentrations were usually elevated within the dumpsite whereas concentrations of *C. perfringens* were not. As mentioned earlier, *C. perfringens* spores may survive at least 300 yr and are, therefore, excellent conservative tracers of past as well as recent sewage pollution (Duncanson et al., 1986). Because survival times of fecal coliforms in marine environments are relatively short, ranging from hours to weeks (Gunnerson, 1988), their presence would indicate only relatively recent sewage contamination (Gordon, 1972; McFeters and Stuart, 1972; Verber, 1976; Sawyer, 1988). But longer survival times may occur in organically rich matrices such as "tar balls" (Katz, unpubl.).

Following the temporal pattern expected as a consequence of their reduced longevity in the marine environment, total and fecal coliform concentrations in surficial sediments at NY6, R2, and NY11 decreased markedly after inputs of sewage sludge ceased (Fig. 9).

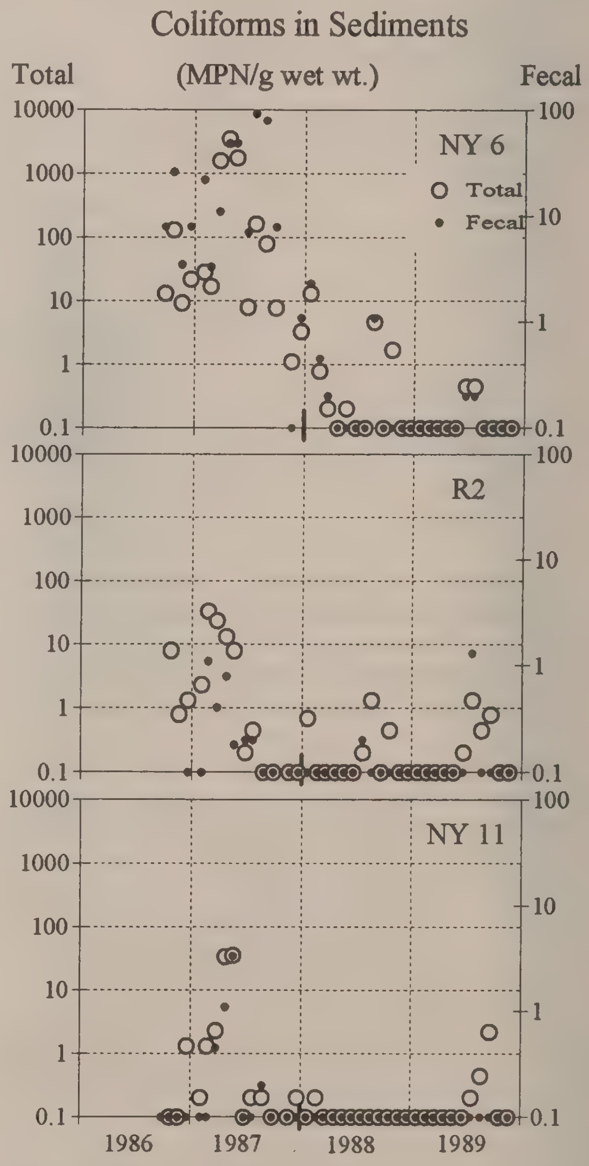

Figure 9

Changes in the concentration of fecal and total coliforms in surficial sediment at replicate stations NY6, R2, and NY11 during the reduced dumping (1986–87) and post-dumping (1988–) periods. Data provided by W. Watkins, US FDA, Davisville, RI.

The temporal patterns for these two sewage indicators were parallel. Moreover, their rates of decrease were considerably faster than the declines in *C. perfringens* abundance (see Fig. 9 and Fig. 4). Total coliform levels approached 10,000 g^{-1} and fecal coliforms approached 100 g^{-1} at station NY6 during the reduced dumping period. A few months following cessation, many of the observations were near or below the detection limit (<0.2 g^{-1}). Similar relative decreases were observed at R2 and NY11, although the detection limit was reached earlier, during the reduced dumping period. The trend at station NY6 also suggests that coliform abundance decreased in response to the reduced volumes of sludge dumped during 1987 (Fig. 2).

It is also important to note that while sludge dumping was occurring, total and fecal coliform counts decreased more rapidly with distance from the dumpsite than did *C. perfringens* counts. Coliforms decreased approximately 100- to 1000-fold from NY6, near the dumpsite, to NY11, 11 km away. This spatial gradient is quite similar to that observed during the 1971 survey described above and contrasts sharply with the relatively modest 30-fold decrease in *C. perfringens* over this distance. Again, these comparisons demonstrate that *C. perfringens* is a more conservative, persistent tracer of sewage contamination.

The few positive, but very low, coliform counts (total and fecal) observed after 1988 may indicate prolonged survival of coliforms in extremely organically enriched microenvironments present in the Christiaensen Basin. Alternatively, these spikes may indicate episodic sewage pollution from the estuarine plume, from vessels using the NY Harbor, or from the mobilization of recently dredged estuarine sediments deposited at the dredge material dumpsite. Resolution of this issue is not possible with the data at hand. Analysis of satellite images of turbidity (Fedosh and Munday, 1982) and the inshore-offshore salinity gradient off Long Branch, NJ (see Introduction to sediment processes), indicate that the estuarine plume closely hugs the NJ coast as it enters the NY Bight. Nevertheless, since it was observed at or beyond the 12-mile site 12% of the time, neither the plume nor the other sources mentioned above can be ruled out as the cause of the coliform spikes.

Inputs to the NY Bight of coliforms from the Hudson–Raritan Estuary are estimated to be more than 100 times the inputs released by dumping of sewage sludge (Mueller et al., 1976). However, the rapid reductions in both total and fecal coliforms and the decreases in concentrations of *C. perfringens* spores in surficial sediments, following abatement of sludge dumping, provides an *a posteriori* demonstration that sewage sludge dumping had a disproportionately large influence in the contamination of sediments in this area. Apparently, the estuarine plume played a relatively minor role in the microbial contamination of the Christiaensen Basin. The persistence of elevated concentrations of *C. perfringens* spores, several years following the cessation of sludge dumping, and their strong signature in buried sediments in the Christiaensen Basin, demonstrates that sewage sludge strongly dominated the chemical contamination of these sediments. Metal distributions support this interpretation. Zdanowicz (1991) observed the highest lead values in buried sediments immediately adjacent to the northwest corner of the sludge dumpsite, where most of the sludge dumping took place. During dumping, elevated levels of lead were also found in surficial sediments over the sewage dumpsite, reflecting recent sewage inputs. In contrast, analyses of metals in the buried sediment (5 cm stratum) revealed a general distribution pattern centered over the basin and relatively low lead concentrations in the sewage sludge dumpsite. Thus, during dumping, a number of indicators of sewage sludge contamination were found in the fine-medium sands (Freeland et al., 1976) within the dumpsite, but because this area did not efficiently sequester fine particulate organic matter and associated contaminants, only moderate concentrations of contaminants were incorporated into the buried sediment layer. On the other hand, the depositional nature of the Christiaensen Basin and its proximity to the sludge dumpsite resulted in efficient retention, concentration, and incorporation of organic matter and associated sewage sludge contaminants into the muddy sediments in the basin.

Forecast

Background concentrations of *C. perfringens* in surficial sediments from the outer NY Bight continental shelf, outside the Hudson Shelf Valley, or from Georges Bank are 10–20 spores g^{-1} (Cabelli and Pedersen, 1982). Data from the present study suggest that the "recovery time," i.e. the time when the highly sewage-contaminated surficial sediments in the Christiaensen Basin reach these levels, will be 15–20 years after the closure of the site (Fig. 4). This is obviously a very rough approximation based on an extrapolated period triple the length of the period over which we have data. It remains the task of future monitoring to evaluate the accuracy of this simple forecast based on extrapolation.

Presumably there are several major mechanisms generating the observed reductions in *Clostridium perfringens* spore concentrations in surface sediments following the cessation of sewage sludge dumping. The relative importance of these will vary spatially and temporally. In strongly depositional areas of the Christiaensen Basin, decreases in sediment spore concentrations would result from gradual dilution with cleaner NY Bight apex "background" sediment that is deposited from the overlying water column and would result from episodic burial under less polluted sediments resuspended from the upper Hudson Shelf Valley and flanking areas during storm events (Manning, 1995). In areas characterized as weakly depositional, where the accumulation of moderate levels of sludge contaminants during dumping was more related to the proximity to the dumpsite than to a strong net deposition tendency, perhaps the dominant process is the gradual winnowing of fine-grained particulates and associated spores and contaminants. Along the eastern periphery of the basin, where one expects only a relatively thin lens of contaminated sediments to have accumulated during dumping, perhaps storm-related erosion of the entire contaminated stratum will be the dominant mechanism of sediment depuration.

We suggest that the temporal variability in spore concentrations will continue to be large. Major storms will probably scour or remove the "improving" surficial layer at muddy, depositional sites, exposing the relatively higher counts in the subsurface layer and, simultaneously, will export and overlay contaminated sediments onto cleaner sandy sediments found outside the basin and in the Hudson Shelf Valley. These resuspension events would be followed by periods of improvement resulting from deposition and vertical mixing of cleaner sediments into the sediment column. A more strict operational definition of "recovery time" would be when the entire sludge-contaminated stratum, not merely the surface layer, is either exported from the basin or capped by cleaner sediments such that the contaminants are beyond the reach of storms and biological mixing by benthic animals.

Future studies of the recovery of the NY Bight should quantify the vertical distribution and thickness of the sewage-contaminated layer and the dynamic processes operating differentially in the various sedimentary, biological, and topographic regimes. *Clostridium perfringens* spores appear to be a very reliable, persistent, and specific indicator of sewage pollution, useful in such characterizations and in understanding the influences that a variety of sediment processes have in determining residual sewage sludge contamination and the rate of sediment depuration.

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Audience Questions

Question: Is it fair to say that *Clostridium perfringens* are viable in terms of the marine environment; that is to say, is there really any likelihood of them vegetating in the environment?

I. Katz: Yes, these *Clostridium perfringens* spores are viable and can grow under the right temperature, reduced oxygen, and availability of proper food elements. These spores remain viable and can germinate after long periods of time, if the specific conditions are present. However, there is very little likelihood that these vegetative conditions are met in the stressful marine environment at the 12-mile dumpsite.

Question: I had some limited data showing an astronomical rise in 1989 at one station. Your coauthor told me that your group also found very high numbers in 1989, but I did not see it in the presentation. Did I miss something?

I. Katz: No. What we are seeing is a slight decrease in numbers; no particular rise in spore count, especially at NY6. If you notice, the numbers were decreasing.

Question: You showed that at NY6 there was a constant spore count regardless of the depth. How do you interpret that?

I. Katz: That's a good question which I really can not answer in view of our findings.

J. O'Reilly: That is a good observation. The densities of *Clostridium* spores a few centimeters down are nearly identical to what they were in surface sediments during active dumping in 1986. Getting back to the earlier question, the *Clostridium* spores are indicators of recent as well as old sewage contamination. Obviously they are not an indicator of fresh sewage input except for that which might be coming from the estuary. What we are showing here, at least after December 1987, is essentially historical sewage contamination abating over time. We did see some bacterial spikes, not in *Clostridium* but in fecal coliform, in 1988 and 1989. Fecal coliforms are viable anywhere from 2 to 50 days in the environment, and are an indicator of recent sewage pollution.

Question: Did I understand correctly that in most of the cores that you have looked at, the *Clostridium* densities increase with depth?

I. Katz: Yes, that is correct. It occurred at NY11, slightly at A56, R2, and NY6.

Question: And yet you are finding in at least the three replicate stations that, as you uncover this sediment, you are finding decreasing concentrations.

I. Katz: The replicate station samples were taken at 0.5 centimeters, i.e. surface samples. The core samples were taken at one centimeter intervals.

Question: I understand, but as time goes by you are looking at samples which were at depth. If I understand correctly, as you go further into depth, spore counts increase. Yet at least in these three stations you are finding fewer spores.

I. Katz: It matters how and precisely where the samples were collected. I do not know if you can really go back to the exact location twice. The core samples and surficial samples were collected at different time intervals, and what we might be seeing is movement of the surficial layers by ocean currents.

J. O'Reilly: My interpretation of that paradox is that it is giving us some insight into how deep in the sediment layer changes are occurring. It would suggest to me that the changes that Irwin's reported on in the last three years are really very superficial; just the near-surface sediments are being affected. If we were having rapid excavation of surface sediments we would probably see some sort of steady level and not a decrease in spores over time. So the fact that we are seeing a time response, a decrease over time, suggests to me that there is a winnowing and dilution of the surficial layers with background sediments proceeding at a relatively slow rate.

Question: It seems to me that on average, at whatever rate the erosion is occurring, if the generalization is correct that the spore densities increase with depth, you have to see more spores over a time. I wonder if one explanation might be just the variability in sampling, and that the three replicate stations either may not be representative of all the others or perhaps we still do not have a good picture of the average depth distribution.

J. O'Reilly: I think when you see some of the data tomorrow, you may come away with the idea that there is a differential rate of reworking of the sediments. Certainly the top veneer is more vulnerable to resuspension, winnowing and dispersion out of the area than are sediments one or two centimeters down. Many sediment variables show changes in the surface layer and little or no change below surface.

I. Katz: I believe more work has to be done on core samples if you are determining at what point the *Clostridium perfringens* spores are no longer present. That would be another interesting portion of the study.

Question: Do you have estimates of continuing inputs? I notice on the maps there were still high levels to the west of the Christiaensen Basin and other estimates of what might be movement in terms of suspended sediments or in the dredge dumping spot. You may be getting those levels going down at that point if you have continued input. At what point are you going to still see some continuing contribution?

J. O'Reilly: Concentrations in the Basin will reach a point where, in time, they will be lower than perhaps other significant sources of spores to the area, such as the dredge dumpsite. I do not think we are at that point yet. The responses in the Basin, I think, are pretty much in reaction to turning off the sludge valve. We do have a good signal-to-noise ratio to date, but we were not sure whether the changes following cessation of sludge dumping would be masked by other inputs. That is exactly the nature of your question. I do not think we are seeing that masking from other sources because the changes are dramatic.

Question: I want to follow up on the previous suggestion. An alternate hypothesis to explain why you are not getting the increased levels that you would expect on those cores which have a long column is that they are being diluted rather than having an erosional basis. You are getting deposition and dilution of your spores from sediments from outside somewhere.

J. O'Reilly: Certainly dilution as well as resuspension and dispersion are operating, and we are not in a position to say what the relevant importance of each process is.

Changes in Sediment Biogeochemistry Resulting from Cessation of Sewage Sludge Dumping in the New York Bight

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ABSTRACT

Redox potential (E_H), as an indicator of sediment biogeochemical characteristics, was measured monthly at three stations during peak dumping, during the phasing out of dumping, and following the cessation of dumping at an oceanic sewage sludge dumpsite in the New York Bight. The stations were located 1.4, 4, and 11 km from the dumpsite. Measurements of E_H were also made less frequently at stations in the surrounding area. Differences in E_H among these stations varied with season, being greatest in summer. Values at the stations nearest the dumpsite (NY6, R2) were usually lower than values at a reference station (NY11). A significant increase in E_H over time was related to the cessation of dumping. A deterministic computer model of sediment biogeochemistry that considers temperature, dissolved oxygen in the overlying water, and labile organic material successfully emulates the record of E_H at NY6, the station nearest the dumpsite. Using the modeled relationships, changes in sediment biogeochemistry are interpreted as being forced by labile organic material availability. Presumably as a result of reduced loading, the area in which sulfidic conditions were found at 0.5 cm in the sediment (implied by E_H<50 mV) decreased from 77 to 4 km^2 between 1983 and 1989. This exemplifies the substantial changes in chemical conditions to which benthic animals were subjected in the New York Bight in response to the cessation of dumping.

Introduction

After more than 60 years of dumping of sewage sludge at the 12-mile site in the New York Bight apex (Fig. 1), a phased cessation of dumping began in March 1986 when a new site (the Deep Water 106-mile Dumpsite), 190 km east of New Jersey, was brought into use. From June 1986 to July 1987, the dumping rate at the 12-mile site was approximately 70% of the full dumping rate, and by 31 December 1987, all dumping ceased (Fig. 2). The Environmental Processes Division (EPD) of the Northeast Fisheries Science Center instituted a program to document changes in the environmental and biological conditions in and around the 12-mile dumpsite, which might result from the phase out and cessation of sewage sludge dumping (EPD, 1988). Termination of dumping provided a rare opportunity to study a large scale environmental perturbation. This paper reports on the relation between stepped changes in organic material loading to the sediment and biogeochemical responses, using measurements of redox potential as the key chemical variable.

Redox potential indicates the potential of the components of a biogeochemical system to give up or take up electrons. Measured as E_H in a sediment, its value is determined by the sum of the relative electron affinities or contributions of the chemical species present in the matrix (e.g. dissolved oxygen, iron, manganese, nitrate, nitrite, sulfide, carbon dioxide, hydrogen) plus the junction potentials of the measurement system. It may also be viewed as a relative measure of "reducing" verses "oxidizing" conditions. In this study redox potential is employed in the sense summarized by Whitfield (1969), where E_H is used as a descriptor of sediment characteristics to be correlated with chemical measurements rather than an exact physicochemical variable indicating the availability of electrons. In the sense of a descriptor, it functions as an index of the chemical condition of the environment that can be compared with discrete measurements and animal community

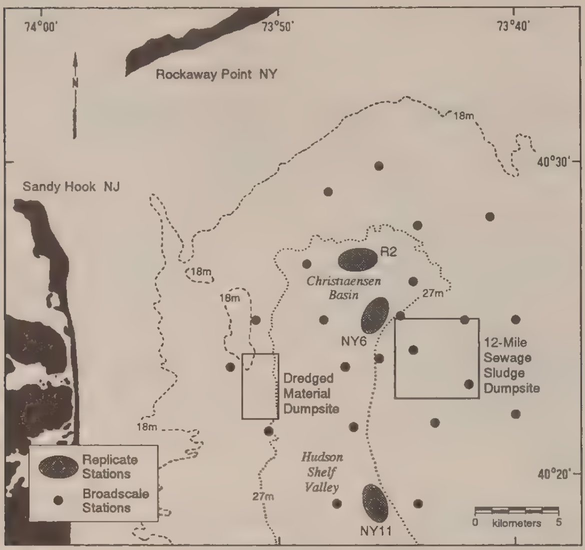

Figure 1
Study area in the New York Bight apex including dumpsites, major bathymetric features, replicate stations (numbered), and broadscale stations.

composition over the period of dumping and during the ecosystem response[1].

This report will concentrate on E_H values measured at three stations during full dumping, during the phase out period, and after the cessation of dumping. These stations were selected to have different organic material loading from the sewage sludge dumping, based on the distances from the designated dumping area (1.4, 4, and 11 km) and differences in water current effects on sludge distribution, deposition, and resuspension. In addition, broadscale observations made throughout the New York Bight apex will be presented to examine the spatial extent of changes related to the cessation of dumping.

At the outset of the multidisciplinary part of the study, a number of statements of anticipated changes were generated (EPD, 1988). This work addresses the following: "The seasonal cycling of redox potential will continue, but the lowest values in the surface sediments (at 0.5 cm) will be ~100 mV higher than current values in the sludge deposition area."

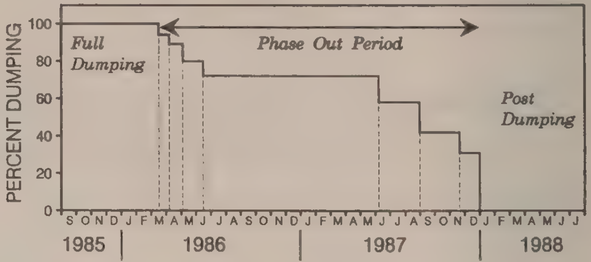

Figure 2
Scheduled phase out of sewage sludge dumping (data from E. Santoro, USEPA, Region II, Marine and Wetlands Protection branch, 26 Federal Plaza, New York, NY 10278. Personal commun., Feb. 1988).

Methods

Sampling

Biweekly sampling for sediment chemistry began at stations NY6 and R2 (Fig. 1) in May 1983. In October 1983, the sampling frequency was reduced to monthly and remained at that level through 1989 (though many of the 1984 and 1985 data were lost in a laboratory fire in September 1985). The multidisciplinary, multi-agency part of this study began in July 1986 and continued through November 1989 at three "replicate" stations (NY6, R2, and NY11) that were sampled monthly. Station NY6 (1.4 km from the dumpsite) was chosen to represent the area in the Christiaensen Basin where sewage sludge was expected to accumulate most heavily; R2 represented an enriched site in the Basin but removed (4 km) from the dumping area; and NY11 was considered a reference station, 11 km south of NY6. These sites on the eastern side of the Hudson Shelf Valley were selected to avoid, as far as possible, any effects from the dredged material dumpsite. Possible influence from dredged material was monitored yearly by occupying closely spaced stations along three well-documented transects (not included in this study; Zdanowicz[2]) between the two dumpsites. Additionally, 21 stations surrounding the 12-mile dumpsite and basin were sampled in even-numbered months to provide synoptic broadscale data to determine the spatial extent of any changes (EPD, 1988).

Sediment samples were collected with a Smith-McIntyre grab deployed at 20–25 m/min and retrieved at less than 10 m/min. If the sediment was coarse, upon retrieval the grab was set in a receptacle containing seawater to provide back pressure so that the interstitial and overlying water could not drain. Disturbance of the surface sediment and the overlying water due to the ship's motion was minimized by placing four baffles constructed of 7-cm diameter core liner in the grab immediately after it was on board. The baffles were inserted approximately 2 cm into the sediment and extended approximately 4 cm above the sediment surface.

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[1] The distinction is made between redox potential as the theoretical measure of the electron demand or supply of a matrix compared with E_H which is the potential actually measured in a sample. E_H includes junction potentials, effects of poisons, etc. and is expressed relative to the hydrogen electrode. Hereafter, for purposes of style, I will use the two interchangeably with the latter meaning of E_H.

[2] V. S. Zdanowicz, NMFS Sandy Hook Laboratory, Highlands, NJ 07732, pers. comm., 1988.

Redox Measurements

Four vertical profiles of redox potential (measured as E_H) were taken in each grab within one hour of sediment collection using a Fisher Scientific model 640 portable pH–millivolt meter. The instrument was calibrated to within 1 mV at 10, 100, and 1,000 mV using a Cole-Palmer pH–mV calibrator (#5657–10). A platinum electrode (Thomas Scientific #4096-D20) with a band of platinum 6 mm diameter × 4 mm height was used as the sample electrode. Depth in the sediment was measured to the center of the band except for the "zero" reading which was made with the band just immersed in the sediment. Measurements were made below the sediment surface at 0.0 cm, 0.5 cm, and at one cm intervals from 1.0 through 10 cm below surface. The reference electrode was a Fisher Scientific (#13–639–62) sleeve junction calomel electrode. This system was calibrated using three $K_3Fe(CN)_6$-$K_4Fe(CN)_6$ solutions of differing redox potentials (ZoBell, 1946; Orion[3]). Before measuring each profile, the electrode was first equilibrated with the sediment at 10 cm in the grab, then with the overlying water. Readings at each depth interval were accepted when the rate of change was less than 1 mV in 10 s. The average time to stability was about 85 s but varied from essentially instantaneous to more than 5 min. Measured mV readings were corrected to the hydrogen reference electrode scale by adding 244 mV (Bagander and Niemisto, 1978). Each value reported is the average of 12 measurements, four E_H values from each of three grabs.

Labile Organic Material Analyses

Estimates of labile organic carbon in sediments were made by adding a 0.1 cm^3 portion of sediment or (in the standardization) 1 mL of D-glucose solution (0.42 to 1.26 mM) to one bottle in each pair of 3 sets of 300 mL biochemical oxygen demand (BOD) bottles which had been simultaneously filled with surface sea water. The seawater in the other bottle of each pair was left untreated and served as a blank. Samples and blanks were then incubated for 13 d at 20°C and the amount of oxygen remaining was determined by the azide modification of the Winkler titration (APHA, 1971) using phenylarsine oxide in place of thiosulfate (Kroner et al., 1964). The amount of oxygen consumed by bacterial oxidation of the D-glucose was used to calculate the labile carbon in the sediment on a "glucose"-carbon basis from BOD_{13}s where BOD_{13} is the biochemical oxygen demand measured on samples incubated 13 d in the manner of the standard BOD_5 (APHA, 1971). The value for r was found to be 0.74 mmol DO_2/mmol glucose-carbon. This may be compared to a theoretical Chemical Oxygen Demand ratio of unity and is nearly identical to the 13-d recovery (75%) of long-term incubations (130 d). Formaldehyde-poisoned controls for station NY6 averaged less than 8% of the value for companion replicates that were not poisoned. Further, the difference between poisoned sediment and poisoned water-only determinations was negative in over 30% of the determinations. Given this small effect, values were not corrected for what is probably random experimental error below the practical limit of detection.

To estimate the carbon derived from settled phytoplankton material, the concentration of major plant pigments in sediment was also measured. A 1 cm^3 portion of sediment was placed in a glass vial and frozen until analyzed. Pigments were extracted from sediments overnight with 100% methanol, diluted to 50% with water; the chlorophyll was then concentrated on a C_{18} column and eluted with 1.8 mL methanol. The extracts (20.0 mL injection volume) were analyzed by isocratic High Performance Liquid Chromatography using methanol pumped at 1.0 mL/min. A Kratos Spectroflow 980 fluorescence detector was set for an excitation wavelength of 430 nm and the emission measured at a wavelength of 550 nm. Methanol-based standards were prepared using chlorophyll *a* obtained from Sigma Co. Standards were kept frozen until used and analyses were performed within 1 week after extraction during which time all material were maintained in the dark at <4°C.

Statistical Procedures

A statistical technique of pseudo-replication over time (Pikanowski, 1995) was used to test whether a variable responded to the change in treatment (sewage sludge dumping). For a given variable, differences were calculated between the value at the treatment station and a reference station. The differences before the step change in treatment were then compared with differences after the change. In practice this involved subtracting redox values measured at station NY11

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[3] Orion. 1983. Instruction manual, platinum redox electrodes. Orion Research Inc., Cambridge, MA.

(reference) from those at stations NY6 and R2 (treatments), then comparing the dumping period (1986–1987) with the post-dumping period (1988–1989).

Results

Values of E_H measured at stations NY6, R2, and NY11 at 0.5 cm depth in the sediment are portrayed in Figure 3. Data from this depth are presented since they encompass the largest seasonal, station-to-station, and long-term variation and because 0.5 cm is the depth for which a computer model simulation has been run.

The data for January to September 1985 from station NY6 and all data prior to October 1985 from all other stations were lost in a laboratory fire. However, the major features were noted (Draxler, unpublished) and the general shape of the curve was known from hand plotting of the data as they were collected. Consequently, E_H during this period is portrayed as a dotted line.

During each survey, with few exceptions, the E_H was lowest at station NY6 and highest at NY11, with values intermediate at R2 (Fig. 3). The lowest values at station NY6 occurred in the summers of 1983 through 1985, during full dumping, when E_H values were typically depressed for several months. In the summers of 1986 and 1987, E_H was less than 0 mV during only one month. The highest value observed at NY6 was measured in February 1989, 13 months after dumping stopped.

The lowest value at R2 was recorded in October 1985 (the first value after sampling was interrupted). The second lowest value occurred in the summer of 1986 during the phase out period. The highest value was observed in March 1989. The lowest value at NY11 was recorded in the summer of 1986 and the highest in January 1989.

The amplitude of the seasonal cycle in E_H at both NY6 and R2 appeared to have diminished in response to moderate reductions (30%) in dumping. There was a further decrease in 1988 and 1989 after dumping had stopped; however, the role of unmeasured natural variations in amplitude cannot be totally excluded.

Results of the pseudo-replication-over-time statistical analysis (Pikanowski, 1995) show that the redox potential at 0.5 cm in the sediment at station NY6 changed significantly ($\alpha<0.001$) as a result of the cessation of dumping but that at station R2 it did not ($\alpha<0.6$).

The lowest concentrations of biologically labile carbon in surface sediments were found in winter and the highest values were observed during the warmer months, although sporadic maxima were found at different seasons in different years (Fig. 4). There was a steady decline in labile carbon concentration after dumping stopped, with the lowest values of the study observed in the final year sampled, 1989. Labile carbon, including episodic, short-lived spikes, was strongly associated (r^2=0.27, n=39) with variation in sediment chlorophyll *a* content presumably due to the deposition of phytoplankton material. Depositional spikes can be seen as being superimposed on an underlying "baseline" sediment labile carbon availability. This baseline of labile carbon concentration decreased from about 140 mmol/dm^3 during dumping in 1984–1987 to about 60 mmol/dm^3 in 1989 following cessation. Dissolved oxygen concentrations reached 77 mM (2.4 mg/L) in 1989, the lowest value since the beginning of sludge volume reductions in 1986 (Arlen et al.[4]). In

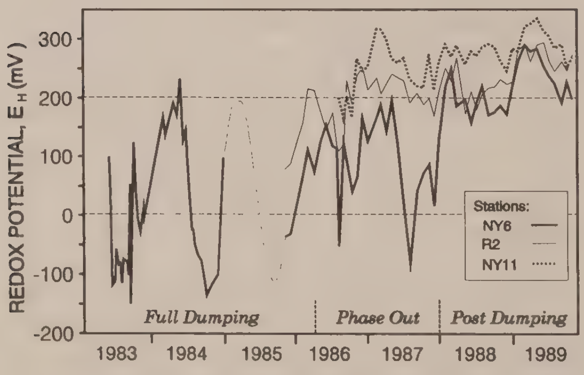

Figure 3

Redox potential at 0.5 cm in sediment at three replicate stations in the New York Bight. Measurements from 1985 were lost in a laboratory fire. The general shape of the distribution was known (see text), so 1985 data are portrayed as a smooth (dotted) line.

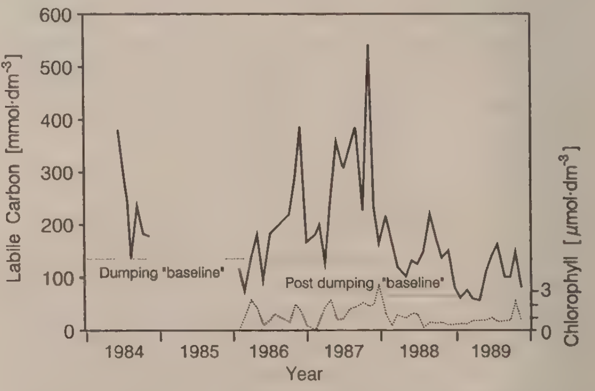

Figure 4

Labile carbon in sediment at station NY6 including "baseline" levels during full dumping and following cessation (see text for further explanation). The solid line is labile carbon (left axis) and the dashed line is chlorophyll *a* (right axis).

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[4] Arlen, L. A., A. F. J. Draxler, and R. A. Bruno Jr. 1994. Hydrographic observations in the bottom water of the New York Bight at the "12-mile" dumpsite: 1983–1990. Unpubl. manuscr., Northeast Fisheries Science Center, National Marine Fisheries Service, NOAA, Highlands, NJ 07732. (Submitted for publication as U.S. Dep. Commer., NOAA Tech. Rep. NMFS.)

contrast, dissolved oxygen minima at NY6 had not been below about 125 mM during 1986–1988; however, before reductions in sludge dumping, values below 15 mM were observed in summers of 1983–1985. The intense near-bottom dissolved oxygen gradients that characterized the full dumping period (prior to 1986) were not observed following cessation (Mountain and Arlen, 1995). The 1989 minimum is within the range predicted in the Plan for Study (EPD, 1988) for post-dumping conditions and probably reflects a general lowering of dissolved oxygen levels throughout the apex due to water column processes (Swanson and Parker, 1988).

Discussion

Factors Influencing Sediment Biogeochemistry

The primary contributor to poising the redox potential in sediment near the sediment–water interface, directly or indirectly, is dissolved oxygen. Major factors governing the depth of penetration of oxygen from the overlying water into a sediment profile include 1) sediment porosity, 2) the rate of sediment microbiological metabolism, 3) the bottom water oxygen concentration gradient, and 4) bioturbation (Walker, 1980; Jahnke et al., 1982). Finer-grained sediments, higher microbial rates, and lower oxygen concentrations in the overlying water result in the lower boundary of the aerobic zone occurring deeper in the sediment–water profile. The seasonal movements of one biogeochemical reactant, dissolved oxygen, are illustrated in Figure 5. In the same way that the above factors affect dissolved oxygen penetration, they also affect the vertical position of concentration maxima of other chemical species, resulting in a continually changing array of oxidized and reduced pairs throughout the sediment profile. The presence and position of chemical species determines the redox potential in that strata. Bioturbation also is important in defining the distribution of chemical species (e.g. the introduction of oxygenated water deep into the sediment by the irrigation of burrow tubes). Since it was not measured and is intermittent in space and time, bioturbation is assumed, perhaps incorrectly, to produce higher average porosity in sediments.

Changes Accompanying Phase Out and Cessation

The primary question addressed here is whether the chemistry of the sediment at the most contaminated station (NY6) changed after dumping reductions began in March 1986. Several pieces of evidence from the redox potential data suggest that it has. First, the maximum E_H value observed at 0.5 cm depth in the sediment was recorded well after all dumping stopped (in winter 1989). Second, marked increases in the summer minimum E_H were observed. Third, the length of time E_H remained depressed (<0 mV) changed from several months in full dumping summers (1983–1985), to about one month in 1986–1987, to not at all in 1988–1989. Fourth, the fit between E_H observed in the sediment and E_H calculated in a computer model, which is largely driven by labile carbon loading from both sewage and non-sewage sources, is quite good (see further discussion of model below). The modeled redox potentials follow the pattern of maxima, minima, and periods of depressed E_H observed at the site. Finally, from the statistical analysis, the probability of no change in E_H following abatement of dumping is <0.001. These changes at NY6 parallel changes in trace metals (Zdanowicz et al., 1995), organic contaminants (Deshpande and Powell, 1995), and bacteria (O'Reilly et al., 1995).

Change in the sediment chemistry is suggested at station R2 as well. Similar to NY6, the highest E_H value at R2 was measured in winter 1989. The lowest existing

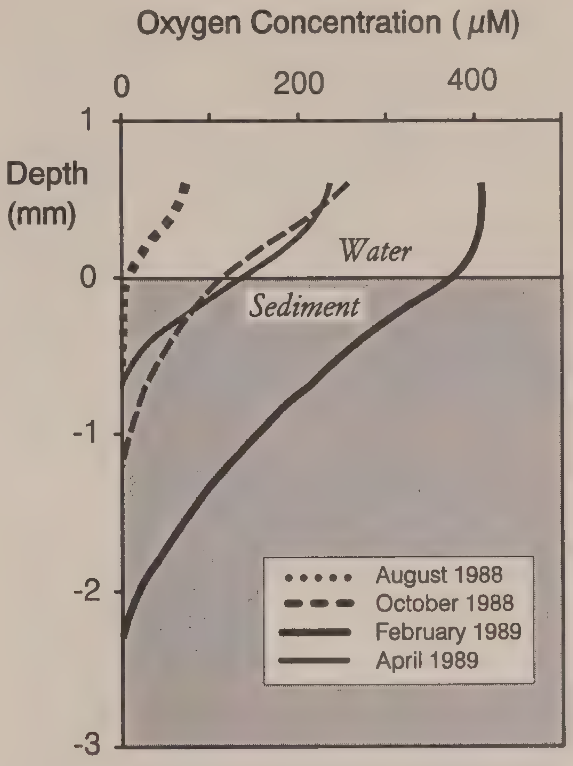

Figure 5
Fine-scale dissolved oxygen profiles in near bottom water and sediment of Long Island Sound (courtesy of J. Mackin and R. Aller, State Univ. New York, Stony Brook, NY).

value for R2 was recorded in October 1985. (Given the seasonal cycling observed throughout the study, the 1985 summer minimum was, presumably, equal to or lower than the October value). Annual summer minima were higher thereafter. This overall trend in summer E_H observations at R2 may also reflect a release from the intense reducing conditions caused by the addition of labile carbon from sewage sludge dumping, as at NY6. However, the statistical analysis (pseudo-replication over time) of R2 relative to the reference station, NY11, indicates that E_H conditions at R2 did not change significantly with respect to the cessation of dumping ($P<0.6$). No significant difference (change) may have occurred if the effects of sludge dumping were essentially equal at R2 and the reference station NY11, if there was an area-wide event during the study, or if seasonal variation masked the overall trend.

Because the statistical analysis (Pikanowski, 1995) requires subtraction of the value at a reference station from the value at each treatment station, there can be only two independent tests using values from three stations. One major assumption of the statistical test is the establishment of station NY11 as a "reference." Presumably, it was not likely to change (due to the treatment) following cessation of dumping. However, redox potential at station NY11 increased between 1986 and 1989 (Fig. 3). An analysis of variance reveals that the mean E_H following cessation was higher than the mean E_H during dumping (Table 1). This brings into question the suitability of NY11 as a reference not affected by dumping. The change at NY11 could be the result of an area-wide change not related to the cessation of dumping or to a wide-area effect of cessation. Neither is verifiable by the statistical analysis. On the other hand, if dumping did alter conditions at NY11, the conclusion that NY6 changed as a result of cessation is strengthened. Nevertheless, the E_H values at the three stations did converge following cessation toward values currently observed at NY11, as would be expected in surface sediments under dissolved oxygen poising and where the deposition of labile carbon does not vary significantly. Such convergence should be, and is, most apparent in winter when bacterial rates are reduced and dissolved oxygen concentrations are increased because of greater water column mixing and solubility. Other evidence comes from elevated levels of *Clostridium perfringens* spores in surface sediments that declined over time at station NY11 and the precipitous decreases in sediment coliform bacteria abundance at all three replicate stations following cessation (O'Reilly et al., 1995). These are further indications that the increase in redox potential at NY11 was due to a reduction of sewage sludge loading, even though the loading at NY11 was at a low level (relative to NY6) in the full dumping period. It follows then that a change across the study area did indeed occur and was related to changes in dumping.

Table 1

Mean and standard deviation of E_H before and after cessation of sewage sludge dumping in the New York Bight apex.

	Before cessation (Jul 1986–Dec 1987)		After cessation (Jan 1988–Nov 1989)	
STATION	Mean E_H,	std. dev.	Mean E_H,	std. dev.
NY11	245	43	286	22
R2	205	37	244	33
NY6	87	75	218	37

Berner (1974) argued that the ecologically important organic material is the "potentially metabolizable carbon," not the commonly measured total organic carbon. The difficulty of measuring labile carbon results in limited data availability, though the concept of biological lability is well established. Lability is a continuous scale from very labile to very refractory. The former includes compounds such as simple sugars which are transported readily across the cellular membrane and enter the metabolic pathways of organisms. The refractory category includes compounds like fulvic substances that have a half-life of thousands of years. Between these extremes, the class of compounds that becomes available in time scales of months may be considered the limit of ecological importance. While the longer-lived compounds in this group may be considered part of the potentially metabolizable pool (at least for coastal environments) and are available on long ecological time scales, because of their slow rate of release they form a background or basal pool upon which are superimposed shorter-term events such as phytoplankton deposition. Berner (1974, 1980) divided sediment organic matter into three classes: G1, with a half-life of a few days; G2, with a half-life of 1 year; and G3, with a half-life greater than 100 years.

Excluding episodic, short-lived spikes, a steady decline in labile carbon concentration was observed after dumping stopped in December 1987 from a "baseline" value of 140 mmol/dm^3 in 1984–87 to about 60 mmol/dm^3 in 1989. There was also seasonal variation (with a summer maximum), presumably from conversion of refractory organic material in the sediment organic material reservoir into a labile form. The transitory nature of spikes of decomposable organic material (having a half-life on the order of a week) is consistent with G1 group material. The negative exponential decline in baseline labile carbon observed between 1987 and 1989 is consistent with G2 material with a half-life on the order of a year.

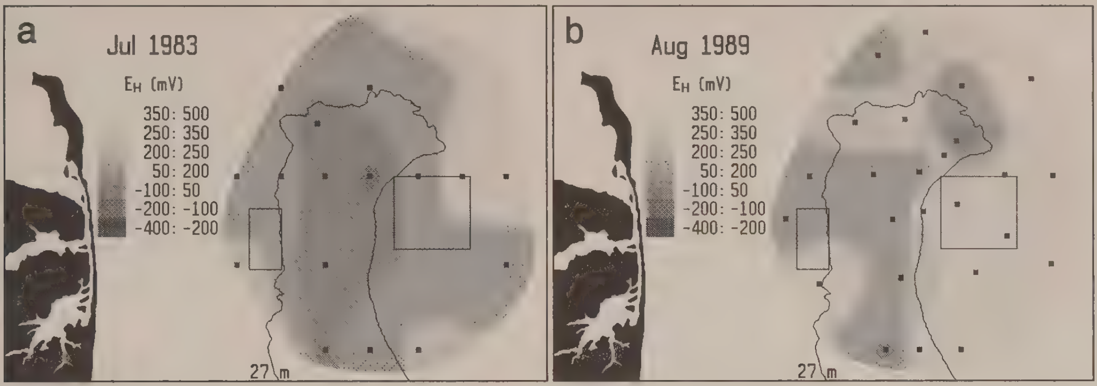

Figure 6
Redox potential at 0.5 cm in sediment for a) July 1983 (full dumping) and b) August 1989 (18 months after cessation of dumping).

Broadscale Changes

The results from the three replicate stations do not address the question of the spatial extent of the control that sewage sludge dumping had on sediment biogeochemical processes in the New York Bight apex. The increased oxidation condition observed at the replicate stations after cessation was evident over a large area. Generally, the near-surface sediments in the deeper parts of the study area (the Christiaensen Basin and upper Hudson Shelf Valley) were more strongly reducing than those in shallower water (Fig. 6a and b). If one considers 50 mV to be an approximation of the highest redox potential at which sulfide is expected, integrating the contoured area having values less than 50 mV (at 0.5 cm in the sediment) for each of 20 broadscale cruises, results in an estimate of the area that was sulfidic on each cruise. That sulfidic area decreased from 77 km^2 in 1983 to 4 km^2 in 1989 (Fig. 7). Though not rigorously testable, this large change parallels in time the reduced loading due to the cessation of dumping, though possible natural variations cannot be ruled out.

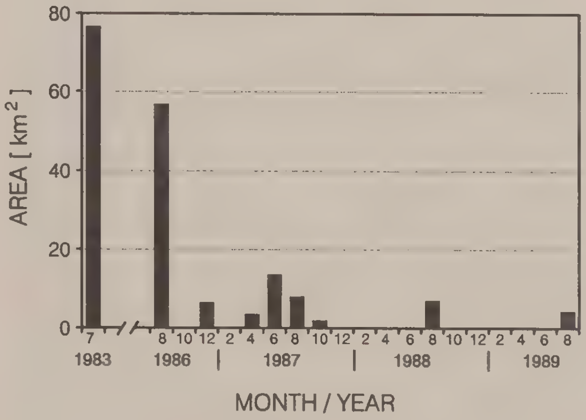

Figure 7
Integral of the area, during 20 broadscale surveys, in which the redox potential was <50 mV at 0.5 cm in sediment. Note that for 11 of the surveys (Oct 1986; Feb and Dec 1987; Feb, Apr, Jun, Oct, and Dec 1988; Feb, Apr, and Jun 1989) no values <50 mV were found.

An idea of how long the buried organic material from sewage sludge dumping will supply energy and, therefore, affect sediment biogeochemistry can be inferred from a graphic extrapolation (negative exponential) based on the area of summer maxima. The estimate will necessarily be imprecise because of variability in the data, as illustrated by the observation that in 1986 the sufidic area in June was larger than in August, while the reverse was true in 1988 and 1989. The analysis suggests that the inferred sulfidic area will disappear relatively rapidly. By 1990, the area was projected to have decreased to <5% of the measured maximum size. Using the annual maxima, the area was projected to decrease to <1% of the 1983 area by 1993. If August values are used, the area was projected to decrease to <1% of the 1983 area by 1996. Consequently, from three to nine years after the complete cessation of dumping, the sulfidic area at 0.5 cm should be virtually absent.

Sediment Biogeochemical Model

Beyond the test of change over time and the descriptive analyses of the experiment, it is desirable to use the data to better understand the mechanisms that relate labile carbon to other sediment biogeochemical variables. One way of gaining insight into these complex processes is to compare field observations with values calculated by models that reflect the interaction of the chemical moieties and to interpret the parameters that cause the models to emulate the actual observations. There are three areas of interest: 1) the relative magnitudes of sources of sediment labile carbon; 2) the control exerted by labile organic matter on sediment biogeochemical processes; and 3) the utility of E_H in following natural sediment processes or, as in this case, those related to environmental perturbations. A deterministic computer model of surficial sediment biogeochemistry (Fig. 8; Draxler, 1988) was developed based on observations conducted during dumping in order to project possible trajectories of the sediment system following cessation (after 1987). This model can be used as a conceptual framework within which to understand the operation of the sediment biogeochemical system and to understand the validity of the observation in this study.

The basic construction of the model was as follows. Input variables (labile organic material loading, temperature, and dissolved oxygen), together with a series of coupled equations relating these variables, were used to calculate the output variables: microbiological activity, standing stock of labile organic carbon, and redox potential (Figs. 8, 9). The model was run for seven years beginning with 1983, using a time step of one month. To simplify results and minimize intra-annual variation that would obscure major trends in projected out-year behavior, the temperature regime for 1983 was repeated for all years. Similarly, two dissolved oxygen seasonal patterns are employed (1983–1985 and 1986–1989) based on summer minima observed to that point in the study. This included minimum summer dissolved oxygen concentrations that declined to 15 mM in 1983–1985 and to 125 mM in 1986–1987. Input values for September were still allowed to drop to 95 mM in the latter period. Carbon loading was assumed to consist of a sewage component (the magnitude of which followed the dumping phase-out schedule) and a non-sewage component (largely from primary production in the water column) of constant magnitude for a model run and equal to a percentage (2–50%) of the carbon loading during the full dumping period. The model was optimized by adjusting constants in the equations that relate input variables in order to obtain redox potential values that fit the patterns observed in 1983–1987. Since the carbon loading through the sediment–water interface and the relationship among the various carbon

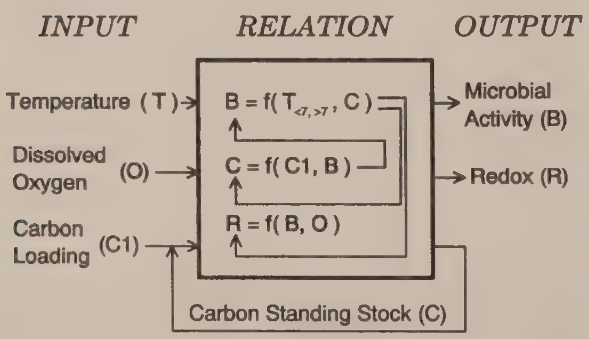

Figure 8
Sediment biogeochemistry model (see text for description of terms).

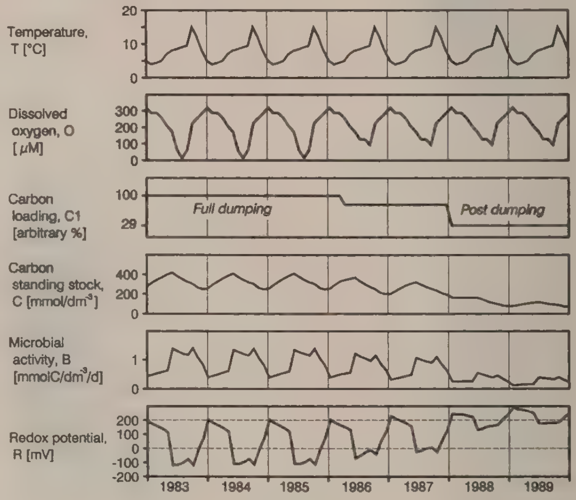

Figure 9
Modeled input and output variables over time for the case in which 29% of the carbon loading (in the period 1983–87) was derived from non–sewage sludge sources.

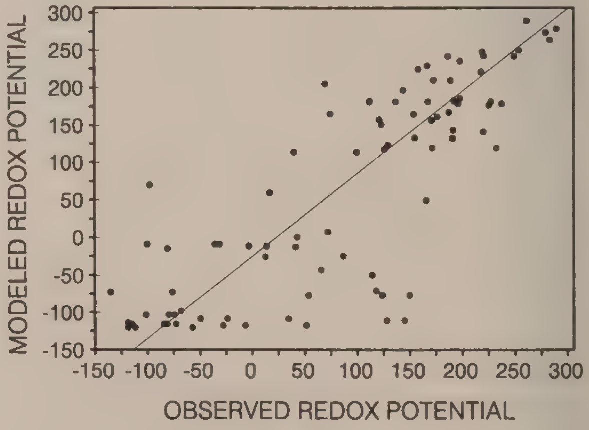

Figure 10
Predicted redox potential from model verses observed redox potential at 0.5 cm in sediment at station NY6 in the New York Bight.

pools were not known at the time of construction, the model was run for five cases of the non-sewage contribution of labile carbon (2, 9, 17, 29, and 50%).

To determine which of these five cases of non-sewage carbon best emulates the field observations, the labile carbon data were examined retrospectively. Lowest biologically labile carbon concentrations were observed in surface sediments at replicate stations NY6 and R2 in 1989. "Baseline" sediment labile carbon content (levels of labile carbon not including phytoplankton depositional spikes or seasonal conversion of refractory to labile carbon; Fig. 4), decreased from roughly 140 mmol/dm^3 in 1984–1987 to about 60 mmol/dm^3 in 1989. This difference may be attributed to the direct plus indirect effects of organic material loading from sewage sludge on the standing stock of labile carbon. If one integrates the area between the labile carbon peaks and the "baseline" as a measure of non-sewage carbon (Fig. 4) and compares that amount to the basal amount (as defined above), then approximately 32% of the measured labile carbon was not from organic matter dumped as sludge during the full dumping period. This is close to the 29% non-sewage run of the model (Fig. 9); therefore, predicted values from this model run may be compared with observed data.

In this comparison of measured with predicted data, model limitations require certain assumptions. Sediment porosity was not included in the model and, therefore, it is necessary to assume that changes in porosity over time do not significantly affect properties and distributions of variables. While the deposition of phytoplankton material from the water column is understood to be episodic, representation as such is beyond the scope of this model. It is, therefore, modeled as a smoothed function over time, reflecting the average labile organic matter input. Similarly, bioturbation is spatially heterogeneous and cannot be included except as a component of the porosity (which is modeled as a smoothed average porosity). Redox potential is then related primarily to microbial rate since the model implicitly includes both seasonal variation through the inclusion of temperature and the activity of major chemical species in the matrix (Fig. 8). Microbial rate is, in turn, modeled as a function of labile organic material availability. Labile organic material availability is considered to be fundamentally dependant on sewage sludge dumping within a temporally varying background of water column input and the conversion of refractory organic material to the labile pool within the sediment. Sediment labile carbon can then be predicted by the model and compared with BOD_{13}-based labile carbon estimates and sediment chlorophyll data as measures of total and non-sewage labile carbon to the benthic system (Fig. 4).

The model simulation for 29% non-sewage labile carbon input calculated that in 1988 and 1989, after the termination of labile carbon input from sewage dumping, redox potential would remain high and the amplitude of the seasonal cycle of redox potential would be small compared with "100%" dumping (prior to March 1986). Since stations NY6 and NY11 are in similar water depths, the expectation was that, with the cessation of dumping of sewage sludge, E_H at NY6 would approach the pattern and magnitude observed at station NY11. This prediction has been borne out as discussed above. The correlation between observed and predicted redox potential is highly significant (Fig. 10; r^2=0.69, n=72), indicating that the model successfully emulates the response of the surface sediment layer at station NY6 to the carbon-forcing regime. Since the input of phytoplankton carbon (chlorophyll) to the sediment was episodic (Fig. 4), and the model was not capable of predicting this, the goodness of the labile carbon representation of the model was tested after correcting for the chlorophyll-related carbon. The correlation between modeled and observed labile carbon was significant (r=0.37, n=39) and the slope of the functional regression (Ricker, 1973; Laws and Archie, 1981) was near unity (0.94), but the amount of variability explained was relatively small (14%). The modeled microbial activity should be related to measurements of seabed oxygen consumption (SOC). SOC was sightly lower during the phaseout period than during full dumping (Phoel et al., 1995). Summer (July, August, September) values declined from an average of 2.7 mmol $O_2/(m^2{\cdot}h)$ in 1985, to 1.7 mmol $O_2/(m^2{\cdot}h)$ in 1986 and 2.2 mmol $O_2/(m^2{\cdot}h)$ in 1987 (1987 is the average of two months as there was no August observation). Following cessation, SOC declined further; to 0.79 and 0.94 mmol $O_2/(m^2{\cdot}h)$ in the summers of 1988 and 1989, respectively. The modeled microbial activity is highly correlated with Phoel's SOC measurements (r^2=0.44, n=51).

The success of the model in emulating observations is consistent with the conclusions that 1) in the area of the dumpsite, about 70% of the labile carbon supply was derived directly or indirectly from sewage sources, while the remaining 30% was from water column processes; 2) the labile carbon from sludge dumping controlled sediment biogeochemical processes over a broad area; and 3) E_H can be used as a descriptor of sediment chemical conditions to which the infauna are exposed.

Conclusions

The dumping of sewage sludge in the New York Bight produced dramatic alteration of sediment biogeochemistry over a large area during the 60 years of the practice. When dumping ended, the sediment system rapidly responded to the decreased supply of labile organic material, which resulted in more oxidizing conditions in surface sediments. E_H (at 0.5 cm) at the sludge accumulation station increased by more than the ≈100 mV increase that was the anticipated change at the start of the study (EPD, 1988). This in turn eliminated the intense dissolved oxygen gradients in the near-bottom water that were observed during the period of full dumping. This study demonstrated how control of the sediment biogeochemistry in the New York Bight is based on availability of labile organic material when account is taken of sediment porosity and seasonal variation in temperature and dissolved oxygen. Redox potential served as an effective proxy variable for discrete chemical measurements, over both space and time, permitting many more measurements than by routine methods. The utility of redox potential as a monitoring tool is not limited to this study. Regulation of sediment biogeochemistry, similar to that observed in the New York Bight, is to be expected in a wide range of benthic environments that experience a seasonal temperature regime and moderate to high labile organic matter loading. This will be reflected in the redox potential and, because of the biological activity of many of the compounds involved (e.g. dissolved oxygen, hydrogen sulfide), has implications for the benthic communities.

Acknowledgments

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Audience Questions

Question: What criteria did you use to discriminate between the labile and refractory carbon?

A. Draxler: Labile carbon was defined as microbially degradable. It was measured by placing a 0.1 cc of sediment in surface sea water and incubating it for 13 days at 20°C. The uptake of oxygen was compared with a paired bottle of sea water that was treated identically.

Question: I notice as a chemist you put the redox on the independent axis, that is to say redox causes the biology. As a biologist, I feel much more comfortable flipping it around—the truth is probably somewhere in between. Do you have any insight into that—the chicken or egg sequence?

A. Draxler: The box diagram was drawn with the physically smaller things at the top so that it may be called a "top down" control model. Traditionally this would be considered "bottom up" control. My feeling is that such things as carbon, sulfide, and perhaps some of the other components of the sediment biogeochemistry have a permitting and perhaps a proactive role in the recruitment and the retainment of benthic organisms.

Changes in Sediment Oxygen Consumption in Relation to the Phaseout and Cessation of Dumping at the New York Bight Sewage Sludge Dumpsite

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ABSTRACT

Seabed oxygen consumption (SOC) was used as a measure of benthic community metabolism to indicate the effects of high organic loading on benthic communities in the vicinity of the New York Bight apex sewage sludge dumpsite (12-mile dumpsite). During the decade between 1974 and 1983 the volume of sewage sludge dumped at the site increased 89%. A concomitant 57% increase in SOC rates ($P<0.05$) was measured at sites in the 12-mile dumpsite and adjoining Christiaensen Basin. Monthly sampling of SOC rates across the dumpsite and Christiaensen Basin were initiated in March 1985 and continued through September 1989. The phaseout of sludge disposal at the site began in July 1986 and was completed in December 1987. SOC rates were consistently elevated at stations which were strongly influenced by sludge dumping but decreased significantly after 1986, during the abatement and after the dumping ceased. By 1988 the SOC rates of those stations which had been directly influenced by dumping had declined to values similar to the least influenced station, east of the dumpsite.

Introduction

The consumption of oxygen by the seabed (SOC) is an important process in benthic ecology. SOC has been used as a measure of benthic community metabolism to understand energy flow and carbon cycling in marine ecosystems (e.g. Teal and Kanwisher, 1961; Kanwisher, 1962; Carey, 1967; Pamatmat, 1971a; Smith et al., 1972; Rowe et al., 1975). Elevated SOC and plankton respiration, coupled with strong seasonal density stratification and sluggish circulation, may cause hypoxia or anoxia in bottom waters (Officer et al., 1984). SOC is also used to indicate the oxidation of organic material and effects of organic pollution on benthic and demersal communities (Baity, 1938; Pamatmat, 1971b; Pamatmat et al., 1973; Smith et al., 1973). Although rates of SOC have been shown to underestimate anaerobic metabolism, which occurs in sediments devoid of oxygen, it is still one of the best measures of integrated biological activity in the seabed (Pamatmat, 1975). Understanding key benthic processes such as SOC is required for the wise use and management of natural resources and protection of the marine ecosystem.

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The New York Bight sewage sludge dumpsite (12-mile dumpsite) is located in the Bight's apex (Fig. 1) and was in use from 1924 through 31 December 1987. The dumpsite is on the east side of the Christiaensen Basin, which is the inshore terminus of the Hudson Shelf Valley (Fig. 1). The basin is generally delineated by the 27 m isobath. The northwest corner of the site received most of the material dumped due to its proximity to the New York/New Jersey ports. The northwest corner is also near the edge of the basin where the flat shelf deepens from 24 m to 34 m (Fig. 2). Diving observations by the senior author have confirmed that sludge, dumped in the site, flows downslope into the depositional Christiaensen Basin.

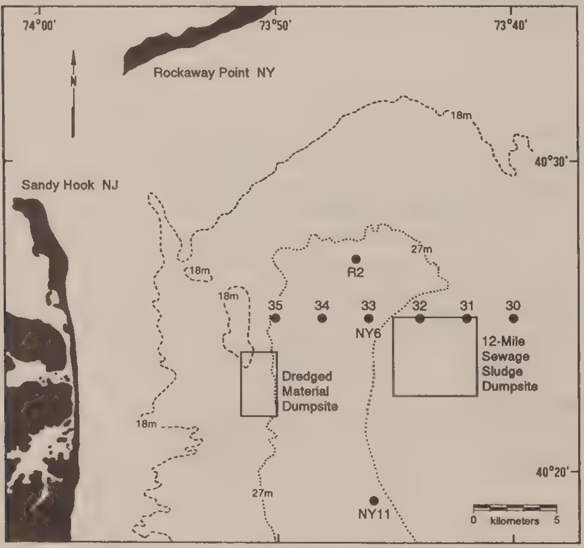

Figure 1

Locations of the 12-mile sewage sludge and dredged materials dumpsites, the six seabed oxygen consumption (SOC) transect stations (30–35), and the three replicate stations (R2, NY6, NY11) in the New York Bight. Station 33 and NY6 represent the same sampling site. The Christiaensen Basin is delineated by the 90-foot (about 28-m) isobath.

The quantity of sewage sludge dumped at the site increased by 89% during the decade of 1974–83 (Phoel et al., 1984), then remained roughly constant at about 1.8 million wet tons per quarter until the beginning of the phaseout in March 1986 (Fig. 3). Nassau County (Long Island, N.Y.) dumped its sludge in the northeast corner of the dumpsite, near station 31 (Fig. 1), and was the first municipality to stop sludge dumping. The county started phaseout in March 1986 and ceased dumping in June of the same year. The numbers of users of the northwestern corner (near station 32; Fig. 1) decreased during 1987 and by 1 January 1988 all sewage sludge dumping at the site had ceased.

Studies of SOC rates and associated variables at the dumpsite and surrounding New York Bight apex began in 1974 (Thomas et al., 1976) and were continued aperiodically until March 1985 when the 12-mile dumpsite study began (EPD, 1988). We then sampled the site monthly through September 1989. Studies by Thomas et al. (1976) revealed that winter (March) SOC rates in the Christiaensen Basin and dredged spoil dumpsite were about 20 mLO_2 $(m^2{\cdot}h)$. The rest of the New York Bight Apex averaged around 20 $mLO_2/(m^2{\cdot}h)$. During summer, when bottom water temperatures were warmer, the SOC rates generally increased and became more variable throughout the apex. At the dumpsite however, the rates increased substantially to about 50 $mLO_2/(m^2{\cdot}h)$ (Thomas et al., 1976).

Based on these earlier studies in the apex and measurements of SOC made in other coastal environments (Phoel, 1982), two hypotheses regarding the anticipated changes in SOC following cessation of dumping were formed at the start of the study (EPD, 1988).

Hypothesis I: With the cessation of sewage sludge dumping, seabed oxygen consumption rates in the dumpsite will be reduced from the high rates of about 30–40 $mLO_2/(m^2{\cdot}h)$ to more natural rates of 9–18 $mLO_2/(m^2{\cdot}h)$.

Hypothesis II: Seabed oxygen consumption rates in the Christiaensen Basin may be reduced by approxi-

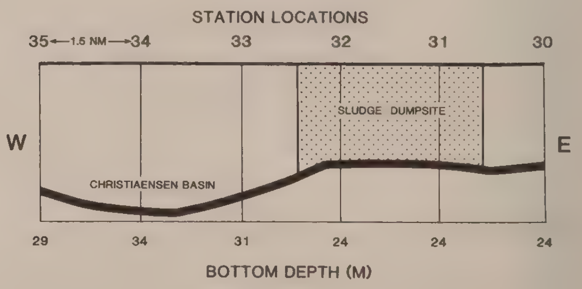

Figure 2

Bathymetry of the seabed oxygen consumption (SOC) transect stations. Bottom depth was measured with a recording fathometer.

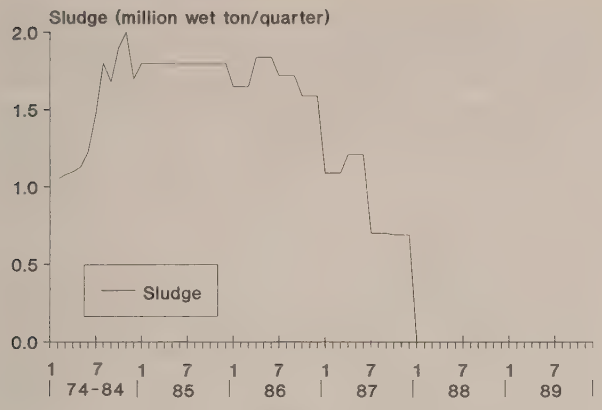

Figure 3

The quarterly rate of dumping of sewage sludge at the 12-mile dumpsite during the period 1974–89. Note that the time axis is compressed for the period 1974–84 (labeled "7484").

mately half, to about 15–20 $mLO_2/(m^2{\cdot}h)$, which is common for accumulation areas if anthropogenic inputs (i.e. dredged materials) remain constant.

The purpose of this paper is to evaluate these hypotheses using the time series of measurements made during periods of dumping, reduced dumping, and no dumping.

Materials and Methods

Sampling Strategy

SOC rates and bottom water temperature, salinity, and dissolved oxygen were measured at six stations on an east-to-west transect across the northern border of the 12-mile dumpsite and Christiaensen Basin (Fig. 1). These measurements were made aperiodically between March 1974 and March 1985, and monthly from March 1985 through September 1989.

Stations were located using LORAN C or more precise navigation systems. All sampling took place within 0.5 km of the station coordinates. Since March 1985, with one exception, the R/V *KYMA* was anchored on station within 30 m of the station coordinates, as determined by LORAN C.

Seabed Oxygen Consumption

A multiple-corer was cast to retrieve simultaneously four relatively undisturbed cores, each having an inner diameter of 5.6 cm and containing 5–15 cm of sediment and overlying water (Pamatmat, 1971a, 1973; Phoel, 1983). Upon retrieval, the cores and overlying water were immediately transferred to constant temperature baths which were equilibrated to ambient bottom water temperatures. Rarely did the incubation temperatures differ by more than ±0.5°C from ambient. SOC data from incubations that differed from ambient by more than ±3.0°C before 1985 and by more than ±2.0°C during and after 1985 were discarded. Galvanic oxygen electrodes, fitted with O-rings for a tight seal, were inserted into the tops of the core tubes. The electrodes were pushed down into the core tubes to expel all air, and a few milliliters of water, via a tube in the electrode fitted with a one-way valve. After about one hour, the electrode was pushed farther into the core tube to expel enough water to rinse and fill a small dissolved oxygen bottle (about 35 mL) the volume of which had been calibrated to 0.001 mL. Dissolved oxygen (DO) concentrations were measured using the azide modification of the iodometric method (APHA, 1965) except that 0.025 N phenylarsine oxide (PAO) titrant replaced sodium thiosulfate (Kroner et al., 1964; EPA, 1974) and proportionately smaller volumes (0.2 mL) of the reagents were used in the 35-mL sample bottles. Dissolved oxygen samples were titrated using a Dosimat model 500. To avoid hypoxia, the cores were incubated only about 6 hours during the warmer months, when bottom DO was relatively low and metabolism relatively high. When bottom water temperatures and associated metabolic rates were low, generally during late winter and early spring, incubations lasted about12 hours. At the end of the incubation period a second DO concentration was measured. Prior to 1985, 4–6 cores per station were incubated; during and after 1985, 8–12 cores from each station were incubated.

During the incubations, DO concentrations were recorded on multipoint stripcharts via the galvanic oxygen electrodes. Examination of the stripcharts revealed that the DO change per unit time was linear in almost all cases. As a result of this linearity the initial and final DO concentrations in the 35-mL oxygen bottles were used to calculate SOC with confidence. The rate of SOC in $mLO_2/(m^2{\cdot}h)$ by each core was calculated using the difference between the initial and final DO concentrations, the volume of incubated water, the cross sectional area of the cores (25.52 cm^2), and the duration of the incubation.

Bottom Water Oxygen Consumption

The measured SOC included oxygen consumption by the sediment and by plankton in the water overlying the sediment in the incubated core. The rate of oxygen consumption in the bottom water was measured so that its contribution to total SOC rate could be determined.

Ten samples, in 60-mL BOD bottles, were obtained from a pool of water collected from water overlying the sediments in unincubated core tubes (about 0.1 m off bottom). Approximately ten additional water samples, in 300-mL BOD bottles, were obtained at each station from a Niskin bottle (about 0.5 m off bottom) attached to the frame of the multiple-corer (see below). Five bottles from each set (0.1 m, 0.5 m) were selected randomly and analyzed for DO concentration immediately following collection. The remaining five samples from each set were incubated in the dark at ambient bottom water temperatures for about12 h when bottom water temperatures were high to 24 h when bottom water temperatures were low. DO concentrations were determined using the modified Winkler method described above and were adjusted based on BOD bottle volumes (calibrated to 0.001 mL). Bottom water oxygen consumption rates ($mL0_2m^{-3}h^{-1}$) were calculated using the difference between mean initial and mean incubated DO concentrations and the duration (h) of the incubation.

Of 660 determinations of bottom water oxygen consumption rates (from both 0.1 m and 0.5 m off bottom), only 10 contributed more than 1% of the SOC rate but less than 3%. We therefore considered the oxygen consumption due to the water overlying the sediments to contribute insignificantly to the total SOC.

Sediment Infauna

At the end of each incubation, the sediment in the core tube was washed through a 1.0-mm sieve. The animals remaining on the sieve were preserved in 10% buffered formalin for later identification, enumeration, and measurement. This analysis provided a rough indication of the macrobenthic invertebrate community and allowed us to consider the possibility that anomalous SOC rates, either high or low, were caused by particular kinds, numbers, or sizes of organisms in the incubated sediments.

Hydrographic Variables

A 5-L Niskin water bottle, attached to the frame of the multiple-corer, was used to retrieve water samples from about 0.5 m off bottom for the determination of temperature, salinity, DO, and oxygen consumption (described above). As soon as the Niskin bottle was on deck, the temperature of the bottom water was measured with a calibrated stem thermometer. Because of the relatively shallow depths (<34 m) and rapid retrieval rate, there was an insignificant amount of temperature change during retrieval.

The concentration of DO in bottom water was measured in duplicate (or more) subsamples (300 mL BOD bottles) from the Niskin bottle using the method described above, except that 0.0375 N PAO was used as the titrant. Salinity was determined using an Autosal model 2100 analyzer.

Statistical Analyses

Since SOC rates showed a marked right skewness, a natural log (ln) transformation was used to bring the data closer to normality. All analyses were conducted on the ln-transformed data. Thus, means and confidence intervals, if reported in the original measurement scale, have all been back-transformed from the ln scale. Statistical analyses were conducted using the GLM procedure (SAS Institute, 1988).

Results and Discussion

General Analysis

A marked decrease in SOC and variability was observed among the six sampling sites following the cessation of sludge dumping (Fig. 4). When data from all stations were pooled, fewer elevated SOC rates were observed following the cessation of dumping than during dumping (Fig. 5). A simple one-way analysis of variance of all data indicated that the mean SOC during dumping was significantly higher than the mean SOC for the post-dumping period (F=27.58, P=0.0001).

SOC and the variability in SOC generally increased with increasing temperature (Fig. 6). (A plot of the untransformed SOC data would show a larger increase in variance at higher temperatures.) Although SOC rates increased with temperature, the response to temperature during dumping was substantially greater than after dumping had ceased (Fig. 6). Multiple regression analysis confirmed that the regression slope of SOC vs. temperature during dumping (0.048) was significantly greater than the slope (0.022) for measurements made after the cessation of sludge dumping (F=4.05; P=0.045). If the input of organic carbon to the sediments was constant and the inputed carbon remained within the dumpsite area, more organic carbon could accumulate during colder months when carbon metabolism was presumably slower compared to warmer months when metabolism was presumably greater (Fig. 6).

The relationship between SOC and sewage sludge loading was explored further using simple linear correlation analyses. The annual mean ln(SOC) was computed for each of the transect stations for the period 1985–87 (when dumping occurred) and for pooled

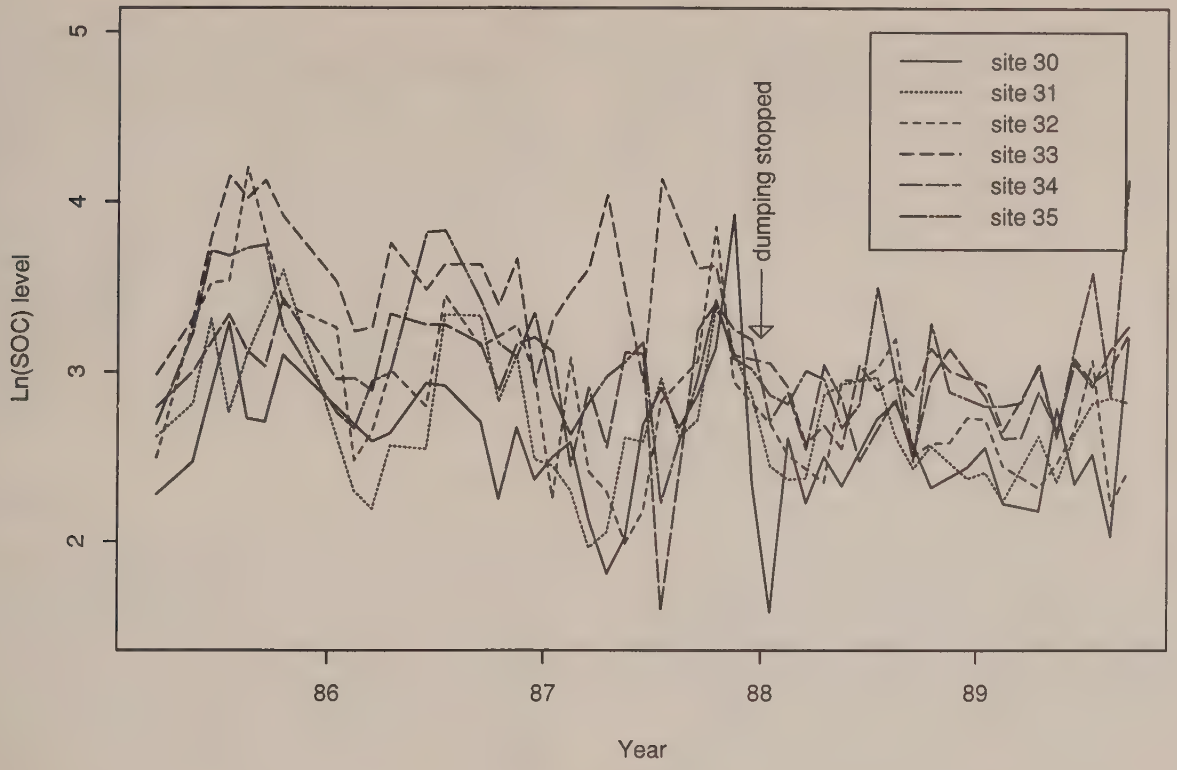

Figure 4
The mean rate of seabed oxygen consumption (SOC) at transect stations 30–35 during the period 1986–89. Replicate SOC values from each station were transformed (ln) before averaging.

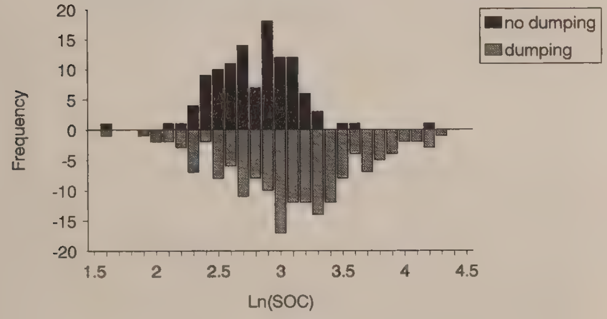

Figure 5
The frequency distribution of ln(SOC) during the dumping and no dumping periods. SOC=seabed oxygen consumption.

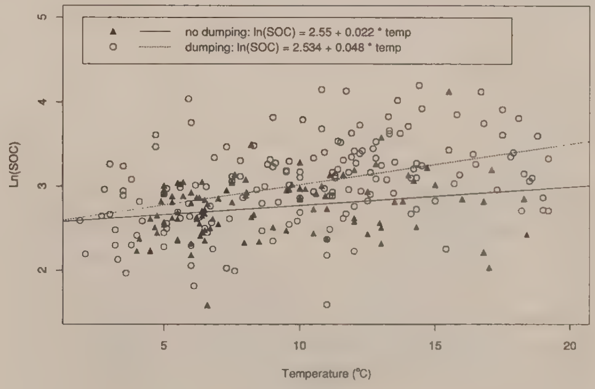

Figure 6
The relationships between ln(SOC) and temperature during the dumping and no dumping periods. SOC=seabed oxygen consumption.

1988 and 1989 data (representative of post-dumping conditions) and correlated with the annual volume of sludge dumped at the 12-mile dumpsite (Table 1). Annual means were computed to minimize the influence of seasonal SOC variability resulting from the annual temperature cycle. The annual mean SOC at all transect stations was positively and highly correlated with the annual volume of sludge dumped. The lowest correlation occurred at station 31, at the northeast corner of the dumpsite. The highest correlation occurred at station 34, in the deepest area of the Christiaensen Basin along the transect, and at station 30, the easternmost station on the transect (Table 1). The high correlation

Table 1

The annual average[1] rate of Seabed Oxygen Consumption, the annual rate of sewage sludge dumping, and their correlation during the period 1985–1989.

		Mean SOC[1] by Station						
Year	Sludge[2]	30	31	32	33	34	35	
1985	1.80	14.5	18.0	29.1	40.0	20.9	27.9	Dumping
1986	1.70	13.1	13.5	18.0	29.5	20.9	23.1	
1987	0.92	12.3	11.7	12.8	30.3	16.6	16.9	Phase Out
1988	0	9.3	11.4	13.0	15.6	14.3	17.2	Post Dumping
1989	0	10.4	11.5	10.6	17.4	16.2	18.2	
R[3]		0.96	0.78	0.83	0.91	0.97	0.82	

[1] SOC rates were transformed (ln) before averaging. Table values represent the antilogarithm of the annual means.

[2] Annual volume of sewage sludge dumped (million wet ton/yr).

[3] R = correlation coefficient for annual mean ln(SOC) versus annual rate of sewage sludge dumping.

observed at station 30 was unexpected. Like station 31, station 30 has a sandy bottom and is located outside the dumpsite boundary at the same depth. This similarity in depth precludes sludge inputs to station 30 from gravity flows. Also, the high correlation at station 30 does not fit the trend of decreasing correlation coefficients observed between stations 34 and 31 (Table 1). These results must be interpreted with care because each correlation coefficient is based on only four values, and factors other than sewage sludge dumping also can affect the SOC rates (e.g. other anthropogenic and natural inputs of organic carbon).

Station 30

During sludge dumping, the lowest rates of SOC (about 9–14 $mLO_2/(m^2 \cdot h)$) on the transect were usually observed at station 30 (Table 1; Fig. 4), located 2.3 km east of the designated dumpsite area (Fig. 1). After dumping ceased, minor decreases in the mean SOC were evident (Table 1; Fig. 7). The extremely high SOC rate in November 1987 remains unexplained (Fig. 7).

From January 1988 through September 1989, the SOC rates, with few outliers, averaged about 10 $mLO_2/(m^2 \cdot h)$ (Fig. 7). This range is close to those measured at other stations with similar sandy sediments under estuarine influence, both within and outside the New York Bight.

SOC rates increased minimally with increasing temperature during summer (Fig. 7). Little or no response to temperature was expected at this station because it was a non-depositional site with sandy sediment; probably the station least affected by the Hudson River plume and furthest from the major dumping site (station 32).

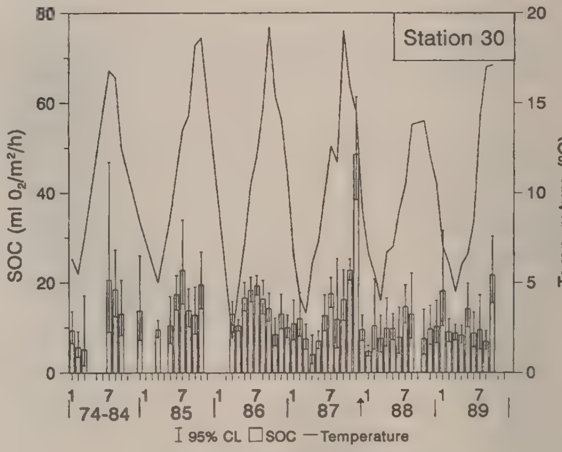

Figure 7

Mean rates of seabed oxygen consumption (SOC) measured at station 30 in relation to seasonal changes in temperature. The means and the 95% confidence intervals are based on ln-transformed data. The arrow indicates the cessation of dumping. The section marked 7484 depicts SOC rates averaged by month from 1974 through 1984. This artificial year is presented to provide an historical background of SOC rates.

Station 31

Station 31 had sediments similar to those at station 30, is 0.5 km inside the dumpsite's eastern boundary (Fig. 1), and was more likely to have received some input of sludge and show signs of recovery when sludge dumping ceased. Sewage plants in Nassau County, N.Y., which serviced primarily suburban rather than urban industrial areas, were the exclusive sources of sludge dumped in the northeast corner of the dumpsite. During dumping, SOC rates averaged about 18 mLO$_2$/(m^2·h) (Table 1; Fig. 8). After the 1986 phaseout of dumping by Nassau County, the SOC rate, with few outliers, averaged about 12 ml mLO$_2$/(m^2·h) through September 1989 (Table 1; Fig. 8). The decrease in SOC rates apparent between 1985 and 1986 (Table 1) coincided with the early phaseout of sludge dumping (about mid-1986) by Nassau County, presumably reflecting the decreased inputs of organic carbon.

From 1985 through 1987 SOC rates increased seasonally with increasing temperatures. After dumping ceased, benthic metabolism no longer followed temperature as strongly but remained relatively constant (Fig. 8).

Station 32

Most sewage sludge dumping occurred near station 32, located in the northwest corner of the dumpsite. Sediments were sandy and similar to those of stations 30 and 31. SOC rates at station 32 during dumping were consistently higher than rates at station 30 or 31 (Fig. 1; Table 1), presumably because the input of organic matter was greater at this station. During and after phaseout, the SOC rates at station 32 declined to or below background levels (Fig. 9). Rates were considered to be background (about15–20 mLO$_2$/(m^2.h)) when similar to rates measured in other coastal areas that were of similar depth, sediment type, and hydrography but that were not receiving sewage sludge, e.g. the coastal areas influenced by the Chesapeake Bay and Delaware Bay plumes (Phoel, 1982).

Before the phaseout of dumping began, the SOC rates were very responsive to seasonal changes in temperature; however, after dumping was stopped SOC rates were not greatly increased during summer when bottom water temperatures were higher (Fig. 9).

Station 33

The highest rates of SOC (about 30–40 mLO$_2$/(m^2.h)) were observed at station 33 during maximal sludge dumping and phaseout (Fig. 10; Table 1). This station lies on the eastern slope of the Christiaensen Basin, just outside the dumpsite's western boundary (Fig. 1), and received large inputs of sludge through gravity flows (Fig. 2). Sediments consisted of soft black mud, similar to stations 34 and 35, also in the Christiaensen Basin. During summer a strong dissolved oxygen gradient, between 0.1 m and 1.0 m off bottom, was present during dumping (Mountain and Arlen, 1995). The gradient was diminished during phaseout and disappeared during the post-dumping period. This suggests that high rates of SOC, as observed during dumping, can deplete near-bottom water of oxygen. After dumping ceased, the rate of SOC decreased markedly, approached background level (about 16 mLO$_2$/(m^2·h)) by 1988, and remained at this level through September 1989 (Fig. 10).

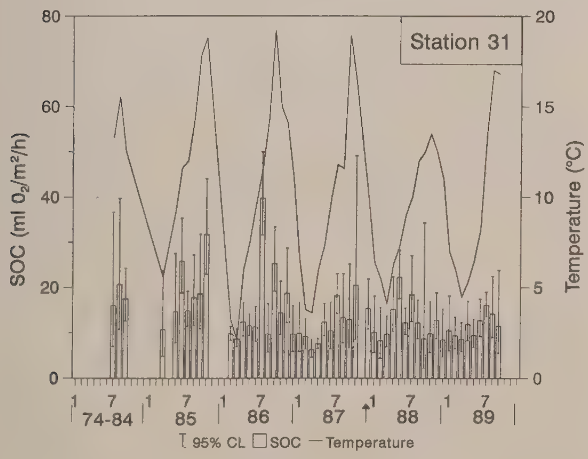

Figure 8

Mean rates of seabed oxygen consumption (SOC) measured at station 31 in relation to seasonal changes in temperature. The means and the 95% confidence intervals are based on ln-transformed data. The arrow indicates the cessation of dumping. The section marked 7484 depicts SOC rates averaged by month from 1974 through 1984. This artificial year is presented to provide an historical background of SOC rates.

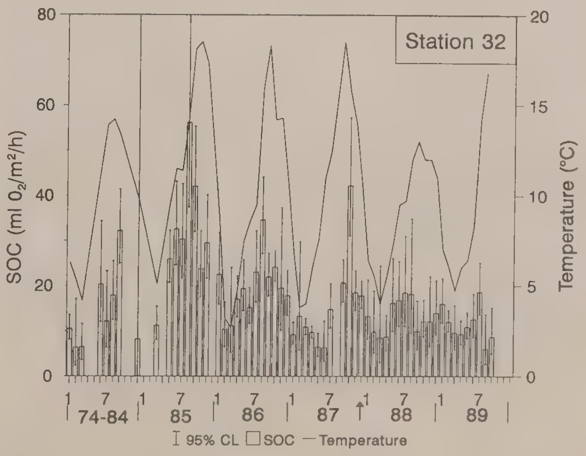

Figure 9

Mean rates of seabed oxygen consumption (SOC) measured at station 32 in relation to seasonal changes in temperature. The means and the 95% confidence intervals are based on ln-transformed data. The arrow indicates the cessation of dumping. The section marked 7484 depicts SOC rates averaged by month from 1974 through 1984. This artificial year is presented to provide an historical background of SOC rates.

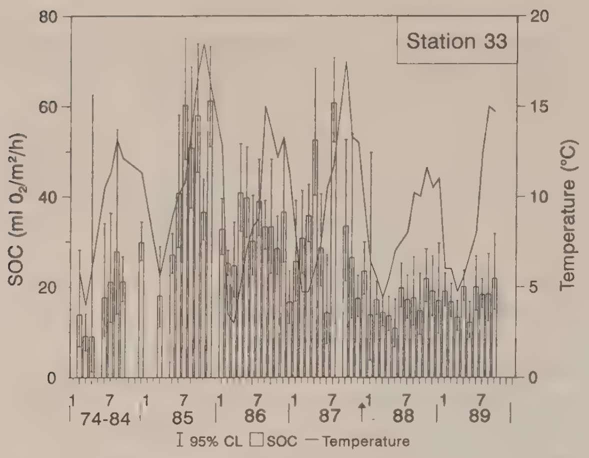

Figure 10

Mean rates of seabed oxygen consumption (SOC) measured at station 33 in relation to seasonal changes in temperature. The means and the 95% confidence intervals are based on ln-transformed data. The arrow indicates the cessation of dumping. The section marked 7484 depicts SOC rates averaged by month from 1974 through 1984. This artificial year is presented to provide an historical background of SOC rates.

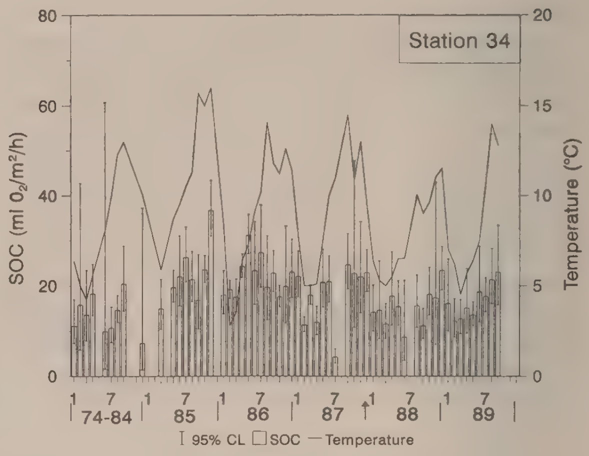

Figure 11

Mean rates of seabed oxygen consumption (SOC) measured at station 34 in relation to seasonal changes in temperature. The means and the 95% confidence intervals are based on ln-transformed data. The arrow indicates the cessation of dumping. The section marked 7484 depicts SOC rates averaged by month from 1974 through 1984. This artificial year is presented to provide an historical background of SOC rates.

Extremely high SOC rates were maintained at station 33 while dumping took place, even when bottom water temperatures were low (although winter SOC rates were still less than the summer rates). After dumping stopped, the rates responded minimally to seasonal fluctuations in bottom water temperature and stabilized at about 16 $mLO_2/(m^2 \cdot h)$ (Fig. 10). Judging from observations while handling the sediments and from remotely operated vehicle videotape, the sediments appeared to become harder and more sandy after dumping stopped. The decrease in the magnitude and seasonal amplitude of SOC following cessation of sludge dumping and the strong correlation between annual mean ln(SOC) and annual sludge loading (Table 1) indicate that sludge dumping probably was the predominant source of organic carbon responsible for the highly elevated rates of SOC at this site.

Station 34

Station 34 is in the center of the Christiaensen Basin and is the deepest station sampled (Fig. 2). It received, through deposition, not only sewage sludge but probably a variety of other natural and anthropogenic inputs to its muddy sediments (Young et al., 1985). However, during dumping, SOC rates at station 34 were lower than those measured at surrounding upslope stations 33 and 35 (Table 1). Post-dumping changes in annual mean SOC rates were the smallest (about 20%) compared with stations 32, 33, and 35 where about a 50% decrease in rates was observed (Table 1). As at other stations, mean annual SOC rates remained fairly constant following phaseout of dumping.

Bottom water temperatures were correlated with SOC rates throughout the study period (Fig. 11). Unlike SOC rates at the previous stations, which ceased or responded minimally to temperature changes after dumping stopped, the rates at station 34 continued to vary seasonally. This suggests that the input of organic carbon remains high and supports higher than background rates of SOC.

Station 35

Station 35 is located on the western side of the Christiaensen Basin, upslope from station 34 (Fig. 2), and 7.5 km west of the sewage sludge dumpsite (Fig. 1). Because of its upslope location it was probably not strongly influenced by sewage sludge dumping. Station 35, however, was about 2.8 km downslope from the dredge spoils dumpsite and received some input of the dredged material (Zdanowicz, 1991). The SOC rates declined significantly between 1985 and 1987, apparently in response to the cessation of sludge and dredge spoil dumping (Fig. 12). This decrease from an annual

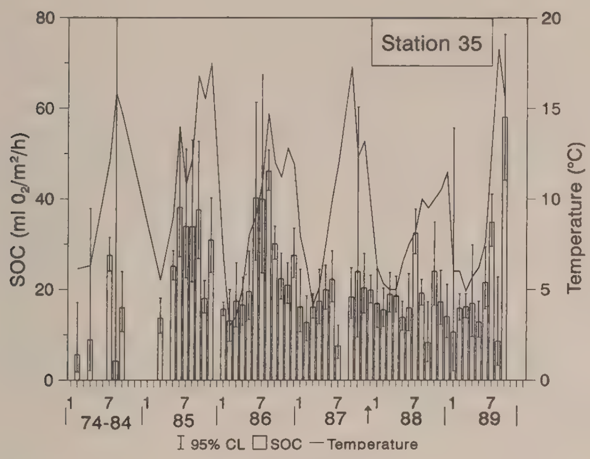

Figure 12

Mean rates of seabed oxygen consumption (SOC) measured at station 35 in relation to seasonal changes in temperature. The means and the 95% confidence intervals are based on ln-transformed data. The arrow indicates the cessation of dumping. The section marked 7484 depicts SOC rates averaged by month from 1974 through 1984. This artificial year is presented to provide an historical background of SOC rates.

mean of about 27 $mLO_2/(m^2 \cdot h)$ to about16 $mLO_2/(m^2 \cdot h)$ (Table 1) was, however, more likely a result of the 75% decrease in dredge spoil dumping in 1985 and 1986 rather than the phaseout of sewage sludge dumping. During the summer of 1989, the SOC rates at station 35 increased to levels measured in 1985 and 1986. This suggests that the cessation of sewage sludge dumping was not responsible for the lowered SOC rates at station 35 in 1987 and 1988 and further implicates dredged spoil dumping as being responsbile. There was an anecdotal report of an increase in the disposal of dredged material in the spring and summer of 1989 (Ingham[1]).

Benthic metabolism rates responded closely to seasonal bottom water temperature fluctuations in 1985, 1986, and 1989 but much less so in 1987 and 1988 (Fig. 12). This suggests that the sediments received less organic carbon in 1987 and 1988 when dredge spoil dumping was reduced.

Conclusions

With the cessation of sewage sludge dumping in the dumpsite, SOC rates at stations 31 and 32 were reduced from summer maxima between 20–40 $mLO_2/(m^2 \cdot h)$ to background rates of about12 $mLO_2/(m^2 \cdot h)$. This magnitude of change, anticipated at the beginning of the study, supports Hypothesis I.

In the depositional Christiaensen Basin, SOC rates were reduced by approximately half, to about 15 $mLO_2/(m^2 \cdot h)$ after dumping stopped. This large decrease supports Hypothesis II in spite of the minimal decrease at station 34 and the confounding influence of the dredged spoils dumpsite on the SOC rates at station 35.

At all stations the correlation coefficients of annual mean ln(SOC) rates vs. the annual volume of sludge dumped were positive and moderate to very strong. This, the decreased amplitude and decline in SOC, and the decreased seasonality in response to temperature, strongly suggest that the decrease in SOC rates was caused by the cessation of sewage sludge dumping.

SOC rates at stations 31 and 32, which were directly influenced by sludge dumping, reached background levels early in 1987 while a quarterly average of 1.0 millon wet tons of sewage sludge was still being dumped. SOC rates at station 33, west of the dumpsite but substantially influenced by sludge dumping, were slower to respond and reached background rates in late 1987, when approximately 0.7 million wet tons was being dumped per quarter. This suggests that the system can endure the impact of about 1.0 million wet tons of sludge per quarter with no increase in SOC beyond the slightly elevated rates typical of other coastal environments that are near estuaries which have high carbon loading but are not directly influenced by sludge dumping.

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Fundamental and Mass Properties of Surficial Sediments in the Inner New York Bight and Responses to the Abatement of Sewage Sludge Dumping*

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ABSTRACT

Surficial sediments in the area of the 12-mile sewage sludge dumpsite in the New York Bight apex are characterized with respect to mineral fraction grain-size, porosity, saturated bulk density, and total organic carbon (TOC). A replication in time (RIT) design was used to test for significant changes in these variables to the cessation of sludge dumping in December 1987. Surface sediment samples were taken for 18 months before and 23 months after cessation at a reference station (NY11) and two sludge-contaminated stations: one (R2) was carbon enriched, the other (NY6), located near the dumpsite, was heavily degraded. The background sediments are poorly sorted, very fine sand (range of means=3.1–3.5ϕ) and a distinctive mud fraction occurs at stations R2 and NY6. Skewed and leptokurtic grain-size distributions at these two stations are an indication of intermittent and fluctuating bottom currents and sediment resuspension events. The degraded station (NY6) exhibited the greatest response to local erosional–depositional events by having the highest temporal variability for almost all the sediment variables measured. There was no significant change in the mean grain-size of the inorganic fraction of the sediment at the three stations after the cessation of dumping. Prior to the cessation of sludge dumping, the sludge-contaminated sediments at NY6 contained a greater organic silt fraction than the less-contaminated sediments at the other stations, resulting in higher mean porosities, lower mean bulk densities (these variables are additive inverses), and higher mean TOC values. After cessation, average sediment porosity and TOC decreased significantly from 68 to 61% and from 5 to 2%, respectively, while mean bulk density increased significantly from 1.4 to 1.6 g cm^{-3} due to dispersion and decomposition of the sludge. Elevated TOC and porosity values and depressed bulk densities occurring at stations R2 and NY6 at the completion of the study are due to the higher percentages of silts and clays present in the background sediments at these sites. Values for the variables measured at the enriched station, R2 are generally intermediate between those of the more heavily degraded station, NY6, and the reference station, NY11.

Introduction

Sewage sludge, that portion of sewage which is collected by settling in sewage treatment plants, had been dumped by barges at a site 12 nmi (22.2 km) from Sandy Hook, NJ, in the inner New York Bight (Fig. 1) since 1924. In April 1985, the U.S. Environmental Protection Agency began closing the site by denying requests for redesignation and ordered that all dumping be terminated by December 1987 (EPD [Environmental Processes Division], 1988).

The sewage sludge dumped at the 12-mile dumpsite was a complex and heterogeneous mixture of organic and inorganic materials. The sludge consisted of 95–

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* Coordinating editors' note: This paper was prepared after the Symposium and therefore not subject to discussion.

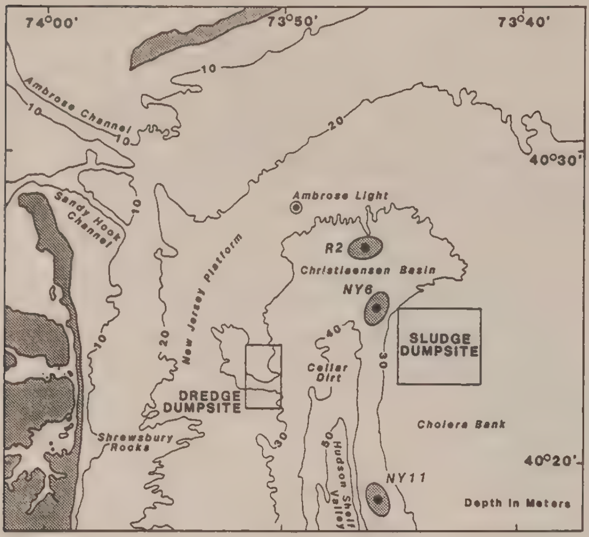

Figure 1

Location of the 12-mile sewage sludge and dredged materials dumpsites and the replicate stations (NY6, R2, NY11) in the New York Bight apex.

99% water with the remaining solids having an average dry density of 1.5 g cm^{-3} (Callaway et al., 1976). Saturated bulk density ranged from 1.0007–1.0181 g cm^{-3} with an average of 1.009 g cm^{-3} (Callaway et al., 1976). About 50% of the sewage sludge was composed of organic matter, the remainder was primarily aluminosilicates resembling soils (Gross, 1972). Campbell et al. (1988) conducted particle size analyses on a 1:1 mixture of sewage sludge in seawater and demonstrated that the largest fraction (60%) is in the clay to medium silt size range (<5.5–4.5ϕ), while the second largest fraction (23%) is composed of solids greater than 3.0ϕ (equivalent to particles larger than fine sands), with most of this material being identified as cellulose fibers and plant fragments. In the New York Bight, suspended particles as large as 3.1 to 3.7ϕ were observed at the sea surface immediately after dumping occurred; these coagulated particles settled after about an hour and left smaller 5ϕ suspended particles in the center of the sludge plume (Jenkins and Gibbs, 1988).

Sludge particles, dispersed into the water column after disposal, eventually became incorporated into the bottom sediments, thereby altering the sediments' physical properties. Knowledge of these fundamental physical properties, which include mineral grain-size and percent total organic carbon (TOC), plus the derived properties of bulk density and porosity, are important not only for assessing the effect of the sludge on the benthic environment, but for a quantitative understanding of local benthic–pelagic coupling.

For example, grain-size reflects the erosional and depositional processes which have created a particular sedimentological environment. The size distribution of the particles which constitute the seabed sediment fabric is most directly the result of the associated hydrodynamic regime (e.g. Ledbetter, 1979, 1984, 1986; Ledbetter and Ellwood, 1980; Blaeser and Ledbetter, 1982; Brunner and Ledbetter, 1989). McLaren (1981) states that the size distribution characteristics of a sedimentary deposit, that is, mean grain-size, sorting (the spread or dispersion of grain-size around the mean), and skewness (the asymmetry of the grain-size distribution curve) are dependent on the processes of 1) winnowing or erosion, 2) selective deposition of sediment in transport, and 3) total deposition of sediment in transport and the grain-size distribution of the source sediment. Other mechanical properties of sediments, such as settling velocity, transportation, permeability, packing, and erodibility, are all dependent upon the sizes of the sediment particles.

Particle size also influences the chemical and biological variables of sedimentary environments. Grain-size is one of the most significant factors controlling the spatial and temporal variability of trace metal concentrations in sediments (Literathy et al., 1985; Horowitz and Elrick, 1988). Grain-size distribution is correlated with benthic invertebrate distribution and often accounts for much of the variability found in the structure of soft-bottom communities (Etter and Grassle, 1992). Thus, Clarke and Green (1988) consider grain-size distribution to be a "nuisance" physical variable with respect to the interpretation of benthic community changes along a pollution gradient. (Nuisance physical or biological variables are variables that are not relevant to the treatment differences being investigated; therefore, to increase precision in the measured response, it is necessary that the nuisance variables be controlled within set limits.) Conversely, the organisms themselves also influence the granulometric characteristics and geotechnical mass properties of their sediment substrate (e.g. Rhoads and Young, 1970; McCall and Tevesz, 1982; Wetzel, 1990).

This paper reports on the effects of cessation of sludge dumping at the New York Bight 12-mile dumpsite on the physical properties of surficial sediments at three nearby sites. These sites differ by the extent to which they have been contaminated by the sludge. Documenting changes in grain-size, porosity, bulk density, and TOC before and after the period in which dumping was discontinued allows assessment of dumping on the sedimentary environment and determination of the response of that environment to the cessation of a major waste input. The results from this paper will assist in the interpretation of contaminant, benthic macrofaunal, and sediment transport data also obtained as part of the 12-mile dumpsite study.

Materials and Methods

Station descriptions and sampling rationale for the 12-mile dumpsite study can be found in EPD (1988), Studholme et al. (1991), and Pikanowski (1992, 1995). The replication in time (RIT) sampling design of Pikanowski (1992, 1995), also known as the BACI (Before/After, Control/Impact) design (Bernstein and Zalinski, 1983; Stewart-Oaten et al., 1986), wherein samples were taken before and after the "experimental treatment" (i.e. the cessation of sludge dumping in December 1987), was used to detect significant changes caused by that cessation of dumping. Three replicate Smith-McIntyre grabs were obtained monthly from July 1986 to November 1989 at each of three "replicate" stations (Fig. 1), which were bathymetrically similar but which represented different levels of sewage sludge accumulation and effects (Pikanowski, 1992, 1995). The station considered to be the most severely degraded by the sludge and containing the greatest accumulations of sludge components or contaminants is NY6 (lat. 40°24.98'N, long. 73°45.58'W; 31 m depth); it is located approximately 1.6 km downslope (west) from the northwest section of the dumpsite where the heaviest dumping occurred. Station R2 (lat. 40°26.77'N, long. 73°46.21'W; 28.5 m depth), on the northern edge of the Christiaensen Basin about 3.4 km north of the dumpsite, is considered to be moderately affected by sludge and biologically enhanced or enriched because of the high biomass and abundance of pollution tolerant species (Reid et al., 1991). The least polluted of the three stations, NY11 (lat. 40°18.92'N, long. 73°45.58'W; 29 m depth), is located on the eastern flank of the Hudson Shelf Valley approximately 10 km south-southwest of the dumpsite center. It is regarded as the reference station, with the lowest levels of sediment contaminants and with benthic fauna that are representative of the upper shelf valley environment (EPD, 1988; Studholme et al., 1991).

Sediment samples were collected from each of the Smith-McIntyre grabs by subsampling with a 3.3-cm diameter plastic core (one core per grab), which was frozen until analyzed to preserve the organic material. After thawing, a 50-ml syringe was used to remove water down to about 1 cm above the sediment surface. Using procedures similar to Robbins and Gustinis (1976), a small portion of the top centimeter of sediment was extracted for porosity, bulk density, and TOC analyses via a modified 5-ml syringe, and placed in a pre-weighed 20-ml glass scintillation vial. The remainder of the top 5 cm of sediment was used for the grain-size analysis.

The particle size distribution of the sediment mineral fraction was determined by means of the standard sieving and pipetting procedures of Ingram (1971), Galehouse (1971), and Folk (1980). The upper particle size-limit chosen for the coarse fraction dry sieving analysis was $\leq -2\phi$ (pebble/granule boundary) and the lower particle size-limit for the fine fraction pipette analysis was at particles $>8\phi$ (clay). The samples often contained high levels of organic material owing to the input of sludge, particularly samples from NY6, and were pretreated repeatedly with several milliliters of 30% H_2O_2 and heated until the organics were digested. It was necessary to remove the organics because their presence interferes with the dispersion of the individual sediment particles during both sieving and pipetting, invalidating the resulting size distributions (Galehouse, 1971; Ingram, 1971). Therefore, because of the removal of organic material, we realize there are obvious limitations in using the inorganic or "background" sediment grain-size data to show pre or post sewage sludge dumping effects. However, it should be noted that all conventional methods of particle-size analyses that include the organic material disrupt and destroy any semblance of the original sediment matrix, thus creating size fractions that are merely the artifacts of the disturbance process. In essence, it is impossible to measure the "particle-size" of the organic matter fraction, because the organic matter forms a continuum within the sediment (Johnson, 1974; Watling, 1988, 1991; Watling[1]).

The grain-size distribution characteristics used to identify sediment trends were calculated using the SEDANA computer program of Bloom et al. (1977) and based on the graphic measures of Folk (1980). The graphic mean diameters (M_z) were determined by averaging the 16th, 50th, and 84th percentiles (Folk and Ward, 1957; Folk, 1980). Other summary statistics included the inclusive graphic standard deviation (SD_I, sorting), inclusive graphic skewness (Sk_I), and graphic kurtosis (K_G), although kurtosis is considered to have limited usefulness in the interpretation of the sediment grain-size distribution (Blatt et al., 1980).

These parameters are often used to determine the direction of sediment transport (e.g. McLaren and Bowles, 1985; Narayana and Pandarinath, 1991; Masselink, 1992), or the source of the sedimentary deposit (McLaren, 1981). However, there has been some discussion on the utility and interpretation of these statistics (e.g. Griffiths, 1967; McLaren, 1981). Knowing the shortcomings of these parameters and to avoid any difficulties, we will refrain from using a cumbrous quantitative approach (as in Blaeser and Ledbetter, 1982). Instead, we will use these parameters as descriptive tools to characterize the sites and make some general and conservative statements about the overlying current regimes. That is, they will be used to make inferences about the local hydraulic climate that would most likely explain the observed sediment distributions and, hence, allow us to speculate on the fate of the sludge. Qualitative specification of sedimentary environments is feasible (Griffiths, 1967); therefore, these parameters will be interpreted in a more qualitative way and used to complement conclusions about sediment transport in the New York Bight apex drawn from past and concurrent studies. It is beyond the scope of this paper to draw any conclusions about the direction of sediment transport or about the source of the material.

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[1] Watling, L. Dept. Oceanogr., Univ. Maine, Darling Marine Center, Walpole ME 04573. Pers. comm., Nov. 1992.

Sediment porosity is the ratio of pore space occupied by the water to total sediment volume and is expressed as a percent (Herdan, 1960). Pore space is calculated as the difference between the total volume of the sediment and the solid material volume. Sediment bulk density, the volume weight of saturated sediment (g cm^{-3}), is controlled by sediment water content and mineral grain density (Lee and Chough, 1987), and its relationship to porosity is as follows (Hamilton, 1970, 1971):

$$BD_{sat} = (P)(D_{pore\ water}) + (1 - P)(D_{mineral\ solids})$$

where BD_{sat} is the saturated bulk density, P is the porosity, $D_{pore\ water}$ is the density of the pore water, and $D_{mineral\ solids}$ is the bulk density of the mineral grains. Saturated bulk density and porosity were calculated using the weight-volume method and are based on the standard techniques of Nafe and Drake (1966), Hamilton (1970, 1971), and Bennett and Lambert (1971). The wet 1-cm^3 subsample was weighed, dried for 18–24 h at 105°C, then cooled to room temperature and reweighed. Following the procedures of Hamilton (1971), a very small salt correction was made to the dry weights of the sediments using the average, rounded salinity for the study of 33 ppt. The correction amounts to an increment to porosity varying, according to porosity, from 0.5–1%. Freezing of the cores prior to analyses may have introduced a source of error in the porosity (Horton et al., 1988), but because the method involves calculated values rather than direct measurement, it is not thought to be critical (Watling[1]).

Analyses for TOC were performed on the 1-ml sediment aliquots taken from the top centimeter of the replicate station cores. The quantity of organic carbon in each subsample was determined on a LECO carbon analyzer using a method similar to that of Kolpack and Bell (1968). A solution of 7% HCl (w/v) was added drop-wise to the sediment samples to dissolve inorganic carbon. When effervescing ceased, one half the initial volume of acid was again added to the sample to provide excess acid for the dissolution of the carbonate. The samples were mixed thoroughly and oven dried at 70°C for 24 h to remove remaining HCl and moisture. The samples were then ground and remixed to ensure homogeneity, and any large remaining fragments (>1 mm) were manually removed. Samples were weighed in crucibles containing approximately 2.5 g copper chip accelerant and combusted in oxygen in the analyzer. The combustion products were dried and filtered into a gravimetric absorption bulb where CO_2 was retained. The absorption of CO_2 was recorded as a weight change in the bulb, permitting the gravimetric calculation of the percentage of TOC in the sediment.

Statistical and data analyses procedures used in the 12-mile dumpsite study, specifically those which utilized the RIT or BACI design, are discussed in EPD (1988) and Pikanowski (1992, 1995). In order to determine if the cessation of dumping can be implicated as an impact on the system, thereby implicating the sludge dumping itself as the disturbance to the system, we tested whether the difference between variable means at the degraded station (NY6), the enriched station (R2), and the reference station (NY11) changed after dumping ceased. This involved calculating the station-to-station differences between the variable means at the three stations for each monthly survey and comparing the means of these differences from before the cessation of dumping (July 1986–December 1987; 18 surveys) to after the cessation of dumping (January 1988–November 1989; 23 surveys). The sampling times themselves are the replicates; sampling at many different times both before and after cessation accounts for the variability due to all sources, including sampling error, spatial heterogeneity and random fluctuations (Stewart-Oaten et al., 1986; Pikanowski[2]). The differences between the means of the two time periods were tested using two-tailed Student's *t* (parametric) and Mann-Whitney *U* (nonparametric) statistics, with nonsignificance (no effect of the disturbance) determined at $p>0.05$. Trends and cycles were not detectable by either parametric (*C* test) or nonparametric (runs test for randomness) tests for serial correlation. Further descriptions of the RIT or BACI design are given in Stewart-Oaten et al. (1986, 1992), Underwood (1991) and Pikanowski (1992).

Results

Grain-size Statistics

The sediments at NY6 ranged from a coarse silt (4.9ϕ) in July 1986 to a fine sand (2.9ϕ) in March 1987 (Fig. 2). However, July 1986 was the exception, and in general the sediments at this station could be characterized

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[2] Pikanowski, R. Northeast Fisheries Science Center, NOAA, Sandy Hook Laboratory, Highlands NJ 07732. Pers. comm., Nov. 1992.

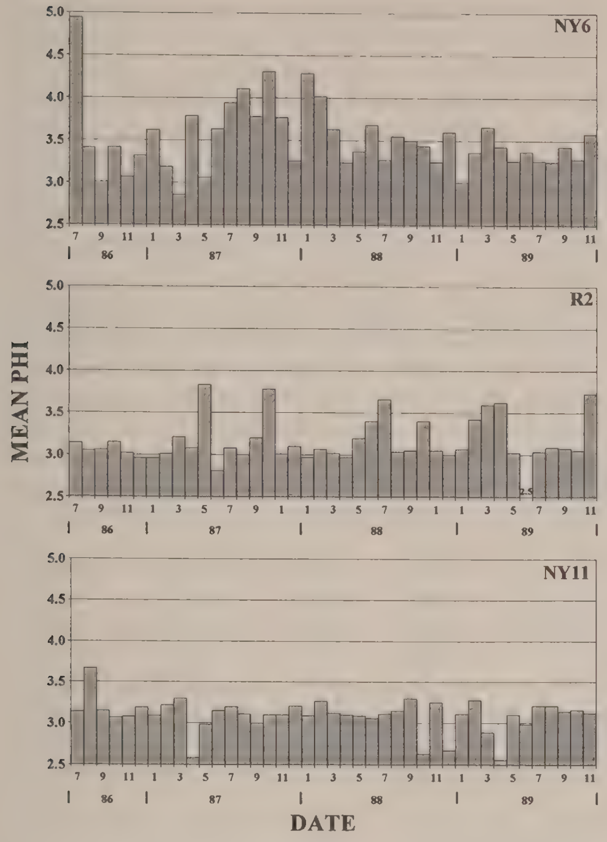

Figure 2
Graphic mean phi (average particle diameter) of sediments at the three replicate stations (NY6, R2, NY11) in the New York Bight apex.

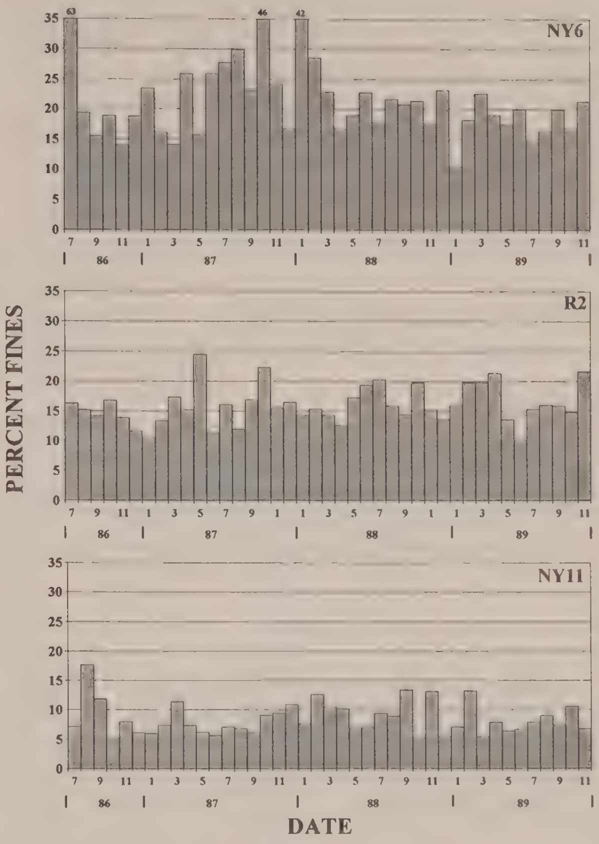

Figure 3
Percent fine fraction (silt plus clay) of the sediments at the three replicate stations (NY6, R2, NY11) in the New York Bight apex.

as a very fine sand (grand mean=3.5ϕ) over the entire study period, with a coefficient of variation (CV) of 11.4%. The sediments at R2 and NY11 could also be characterized as very fine sands (grand mean=3.2ϕ and 3.1ϕ, respectively), but with lower temporal variabilities than at NY6 (CV=8.5% and 6.4%, respectively). Comparisons of mean phi before and after the cessation of dumping within and between stations showed no significant differences (Table 1).

The percent of sediment weight in the fine fraction (silt plus clay) at each replicate station over time is shown in Figure 3. Percent fines at NY6 ranged from 63.4% in July 1986 to 10.4% in January 1989. The large amount of fines in the July 1986 sample is a result of the presence of a dominant (42.9% by weight) clay-size fraction. Overall, percent fines at R2 were lower than those of NY6 and ranged between 24.5% in May 1987 and 10.2% in June 1989. Percent silt-clay at NY11 was lower still, with a range from 17.7% in August 1986 to 5.1% in October 1986.

The inclusive graphic standard deviation indicates that, in general, the sediments at all three stations were poorly sorted throughout the 41-month study period (Table 2). The exception was at NY11 where several samples were moderately sorted because of the slightly lower percentages of both the fine and coarse fractions as compared with the other dates and stations. The presence of pebbles in the March 1987 NY6 and June 1989 R2 samples made these sediments the most poorly sorted (2.0ϕ).

The inclusive graphic skewness at NY6 showed month-to-month variability without a clear pattern (Table 2). The sediment size-frequency distributions ranged from strongly coarse-skewed (negative skewness) to near-symmetrical (near zero) to strongly fine-skewed (positive skewness), with the majority being fine-skewed. The distributions at NY11 were predominately near-symmetrical with less variability over time than at NY6. Size distributions at R2 were all fine-skewed except for the June 1987 and June 1989 samples, which were near-symmetrical and coarse-skewed respectively, due to the presence of a coarse fraction consisting of shell fragments, granules, or pebbles.

The sediment distributions at all three stations were almost uniformly leptokurtic to very leptokurtic (Table 2). The major exception occurs in the July 1986 sample at NY6 which was platykurtic owing to its bimodal distribution and very poorly sorted sediments caused by a large proportion of clay-sized particles and a smaller secondary mode of very fine sand. A few mesokurtic distributions occurred at NY11.

Porosity

Sediment porosity at the three replicate stations showed a decreasing trend in the order of NY6>R2>NY11 (Fig. 4). Porosity values at NY6 ranged from 76.2% in February 1987 to 50.0% in January 1989, with an overall mean of 64.1 ± 6.5% and CV=10.1%. There was less temporal variability at R2 (CV=7.4%) and a lower overall mean of 52.9 ± 3.9%. Sediments from station NY11 had the lowest porosity (grand mean=50.0 ± 3.3%) and the lowest variability (CV=6.7%) throughout the study. Comparisons of pre- and postcessation means between stations; i.e. the means of NY6 to the enriched station R2 and to the reference station NY11, showed highly significant differences (Table 1; t=5.14 for NY6-R2, t=4.13 for NY6-NY11, both $p<0.001$; the positive signs of the t values indicate that the precessation means were higher). However, porosity at station R2 did not change significantly relative to NY11. Comparisons of pre- and postcessation means within NY6 indicates a highly significant decrease in sediment porosity from 68.0% during precessation to 61.5% during postcessation. Porosities at R2 and NY11, however, had slight, but insignificant, increases after the cessation of dumping (Table 1).

Table 1

Results of the RIT technique of Pikanowski (1995) applied to the selected sediment properties, with variable means before the cessation of dumping covering the period from July 1986 to December 1987 and postcessation variable means from January 1988 to November 1989. Differences between means, both inter-station and intra-station, were analyzed parametrically (t-test) and non-parametrically (Mann-Whitney U-test). Symbols, *n.s.*: not significant ($p > 0.05$); *: significant ($p < 0.05$); **: significant ($p < 0.01$); ***: significant ($p < 0.001$).

Property	Station	Precessation Mean ± SE (n = 18 months)	VS.	Postcessation Mean ± SE (n = 23 months)	t-statistic (df = 39)	U-statistic $n_1 = 18$, $n_2 = 23$
Grain-Size (ϕ)	NY6	3.58±0.12		3.47±0.06	0.92n.s	n.s
	R2	3.14±0.06		3.18±0.06	–0.47n.s.	n.s.
	NY11	3.13±0.05		3.07±0.04	0.96n.s.	n.s.
	NY6–NY11	0.45±0.14		0.40±0.07	0.37n.s.	n.s.
	NY6–R2	0.45±0.13		0.29±0.08	1.05n.s.	n.s.
	R2–NY11	0.01±0.08		0.11±0.08	–0.86n.s.	n.s.
Porosity (%)	NY6	68.02±1.26		61.05±1.21	3.94***	336.0***
	R2	51.77±0.83		53.77±0.87	–1.64n.s.	n.s.
	NY11	49.30±0.72		50.62±0.75	–1.25n.s.	n.s.
	NY6–NY11	18.72±1.57		10.43±1.28	4.13***	340.0***
	NY6-R2	16.26±1.43		7.28±1.06	5.14***	358.0***
	R2-NY11	2.46±1.03		3.15±1.08	–0.45n.s.	n.s.
Bulk Density ($g\ cm^{-3}$)	NY6	1.42±0.02		1.57±0.03	–3.97***	337.5***
	R2	1.77±0.02		1.70±0.02	2.98**	314.0**
	NY11	1.84±0.02		1.83±0.01	0.37n.s.	n.s.
	NY6–NY11	–0.42±0.03		–0.26±0.03	–3.29**	324.0**
	NY6–R2	–0.35±0.03		–0.13±0.02	–6.15***	374.0***
	R2–NY11	–0.07±0.02		–0.13±0.02	1.87n.s.	n.s.
TOC (%)	NY6	4.55±0.47		2.33±0.34	3.94***	335.0***
	R2	0.92±0.10		0.93±0.12	–0.06n.s.	n.s.
	NY11	0.26±0.02		0.33±0.02	–2.23*	315.5**
	NY6–NY11	4.28±0.47		2.00±0.34	4.04***	340.0***
	NY6–R2	3.72±0.51		1.41±0.36	3.83***	317.0***
	R2–NY11	0.65±0.10		0.59±0.12	0.37n.s.	n.s.

Bulk Density

Of the three stations, NY6 had the lowest overall saturated bulk density values (grand mean=1.51 ± 0.14 g cm^{-3}; CV=9.3%; Fig. 4). Bulk density was higher at R2 (grand mean=1.73 ± 0.08 g cm^{-3}) and less variable (CV=4.9%). Station NY11 had the highest overall average bulk density values throughout the study period (grand mean=1.83 ± 0.07 g cm^{-3}) and the lowest variability (CV=3.7%). Pre- and postcessation means of NY6

Table 2

Statistical parameters of the sediment grain-size distributions at the three replicate stations. Sorting symbols are M: moderately sorted; P: poorly sorted; VP: very poorly sorted. Skewness symbols are SF: strongly fine-skewed; F: fine-skewed; N: near symmetrical; C: coarse-skewed; SC: strongly coarse-skewed. Kurtosis symbols are P: platykurtic; M: mesokurtic; L: leptokutic; V: very leptokurtic.

	NY6			R2			NY11		
Date	$SD_I(\phi)$ Sorting	SK_I Skewness	K_G Kurtosis	$SD_I(\phi)$ Sorting	SK_I Skewness	K_G Kurtosis	$SD_I(\phi)$ Sorting	SK_I Skewness	K_G Kurtosis
7/86	1.61 P	–0.17 C	0.69 P	1.19 P	+0.24 F	1.51 V	1.05 P	+0.02 N	1.71 V
8/86	1.39 P	+0.16 F	1.70 V	1.22 P	+0.23 F	1.55 V	1.33 P	+0.32 SF	2.22 V
9/86	1.32 P	+0.09 N	1.50 L	1.17 P	+0.24 F	1.54 V	1.21 P	–0.02 N	1.87 V
10/86	1.26 P	+0.16 F	1.82 V	1.32 P	+0.32 SF	1.52 V	0.78 M	–0.25 C	0.94 M
11/86	1.24 P	+0.06 N	1.56 V	1.22 P	+0.21 F	1.57 V	1.15 P	–0.03 N	1.70 V
12/86	1.42 P	+0.13 F	1.59 V	1.19 P	+0.23 F	1.57 V	0.79 M	–0.13 C	1.28 L
1/87	1.59 P	+0.22 F	1.62 V	1.17 P	+0.26 F	1.62 V	0.96 M	–0.15 C	1.35 L
2/87	1.22 P	+0.02 N	1.69 V	1.20 P	+0.24 F	1.55 V	1.02 P	–0.01 N	2.15 V
3/87	2.00 VP	–0.35 SC	2.49 V	1.36 P	+0.32 SF	1.51 V	0.99 M	+0.09 N	2.17 V
4/87	1.44 P	+0.29 F	1.83 V	1.19 P	+0.22 F	1.53 V	1.47 P	+0.04 N	1.37 L
5/87	1.57 P	–0.16 C	2.02 V	1.76 P	+0.45 SF	1.38 L	1.16 P	–0.10 N	1.48 L
6/87	1.49 P	+0.22 F	1.53 V	1.72 P	–0.04 N	2.35 V	0.82 M	–0.13 C	1.24 L
7/87	1.73 P	+0.31 SF	1.33 L	1.24 P	+0.21 F	1.55 V	1.00 P	–0.01 N	1.93 V
8/87	1.55 P	+0.40 SF	1.27 L	1.23 P	+0.20 F	1.62 V	1.13 P	–0.04 N	1.81 V
9/87	1.47 P	+0.32 SF	1.85 V	1.22 P	+0.25 F	1.50 V	1.11 P	–0.11 C	1.44 L
10/87	1.73 P	+0.25 F	0.69 P	1.74 P	+0.44 SF	1.44 L	1.19 P	–0.02 N	1.79 V
11/87	1.59 P	+0.28 F	1.74 V	1.27 P	+0.23 F	1.55 V	1.19 P	–0.04 N	1.77 V
12/87	1.21 P	+0.05 N	1.81 V	1.27 P	+0.28 F	1.52 V	1.11 P	+0.03 N	1.97 V
1/88	1.66 P	+0.32 SF	0.70 P	1.26 P	+0.26 F	1.57 V	1.12 P	+0.01 N	1.67 V
2/88	1.53 P	+0.38 SF	1.38 L	1.20 P	+0.23 F	1.54 V	1.10 P	+0.04 N	2.29 V
3/88	1.42 P	+0.22 F	1.92 V	1.22 P	+0.25 F	1.56 V	1.21 P	–0.02 N	1.80 V
4/88	1.23 P	+0.04 N	1.74 V	1.22 P	+0.23 F	1.58 V	1.20 P	+0.01 N	1.73 V
5/88	1.32 P	+0.12 F	1.75 V	1.32 P	+0.30 F	1.51 V	1.12 P	–0.00 N	1.70 V
6/88	1.57 P	+0.24 F	1.77 V	1.43 P	+0.36 SF	1.48 L	1.16 P	–0.05 N	1.66 V
7/88	1.51 P	+0.03 N	1.84 V	1.60 P	+0.44 SF	1.46 L	1.19 P	–0.01 N	1.82 V
8/88	1.46 P	+0.22 F	1.69 V	1.24 P	+0.25 F	1.54 V	1.14 P	–0.01 N	1.90 V
9/88	1.29 P	+0.19 F	1.86 V	1.21 P	+0.19 F	1.56 V	1.06 P	+0.07 N	2.31 V
10/88	1.41 P	+0.15 F	1.70 V	1.48 P	+0.37 SF	1.47 L	1.07 P	–0.09 N	0.97 M
11/88	1.28 P	+0.06 N	1.68 V	1.23 P	+0.22 F	1.56 V	1.14 P	+0.01 N	2.21 V
12/88	1.46 P	+0.21 F	1.82 V	1.23 P	+0.23 F	1.58 V	1.07 P	–0.01 N	1.01 M
1/89	1.20 P	+0.08 N	1.57 V	1.21 P	+0.23 F	1.53 V	1.09 P	–0.01 N	1.72 V
2/89	1.23 P	+0.11 F	1.83 V	1.49 P	+0.38 SF	1.47 L	1.05 P	+0.07 N	2.10 V
3/89	1.37 P	+0.25 F	2.11 V	1.52 P	+0.41 SF	1.48 L	1.03 P	–0.29 C	1.06 M
4/89	1.17 P	+0.20 F	1.77 V	1.57 P	+0.41 SF	1.44 L	1.42 P	+0.13 F	1.38 L
5/89	1.29 P	+0.08 N	1.67 V	1.19 P	+0.23 F	1.56 V	1.11 P	–0.04 N	1.73 V
6/89	1.41 P	+0.14 F	1.61 V	2.04 VP	–0.18 C	2.39 V	1.16 P	+0.04 N	1.57 V
7/89	1.08 P	+0.01 N	1.90 V	1.22 P	+0.25 F	1.54 V	1.03 P	+0.01 N	2.02 V
8/89	1.17 P	+0.02 N	1.85 V	1.20 P	+0.23 F	1.52 V	1.07 P	–0.01 N	2.09 V
9/89	1.29 P	+0.15 F	1.82 V	1.22 P	+0.21 F	1.54 V	1.10 P	–0.03 N	1.83 V
10/89	1.24 P	+0.06 N	1.85 V	1.21 P	+0.22 F	1.55 V	1.18 P	–0.01 N	1.91 V
11/89	1.51 P	+0.23 F	1.70 V	1.65 P	+0.44 SF	1.44 L	1.10 P	–0.04 N	1.79 V

relative to R2 and NY11 showed highly significant differences (Table 1). There was also a significant difference in sediment bulk density pre- and postcessation within both stations NY6 and R2. Sediment bulk density at NY6 increased from the precessation period (mean=1.42 g cm^{-3}) to postcessation (mean=1.57 g cm^{-3}), while R2 had a significant, but small decrease of 0.07 g cm^{-3} in bulk density after dumping ceased (Table 1). This decrease in bulk density may be due to any number of sampling artifacts or changes in environmental conditions, but is surely correlated with the slight increase in postcessation porosity at R2. Mean bulk density did not change significantly at station NY11 and remained around 1.84 g cm^{-3} (Table 1).

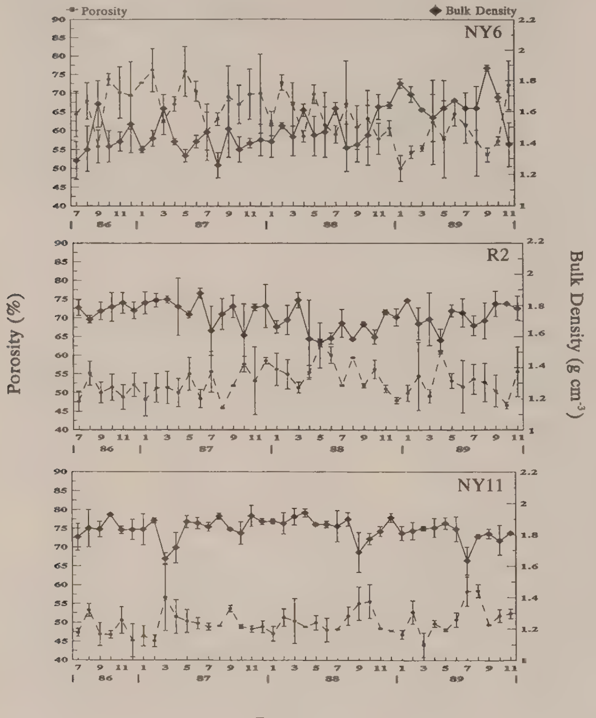

Figure 4
Mean sediment porosity and saturated bulk density at 0–1 cm depth at the three replicate stations (NY6, R2, NY11) in the New York Bight apex. Average n=2±SE for each date.

The additive inverse relationship between porosity and bulk density is apparent when comparing the data for these two variables (Fig. 4). This relationship is further demonstrated through the use of geometric mean regression analysis, the appropriate regression model to use when both X and Y variates are subject to measurement error (Ricker, 1973; Laws and Archie, 1981; Press and Teukolsky, 1992). At NY6 (Fig. 5a), the relationship is linear as expected because bulk density increases as porosity decreases over time (r^2=0.56). The relationship is less obvious at R2 (Fig. 5b; r^2=0.32) because bulk density shows an unexpected significant decrease while porosity is slightly increasing. At NY11, where neither bulk density nor porosity appear to change through time, the relationship is indistinct (Fig. 5c; r^2=0.11). Although the slopes and intercepts are similar among the three stations, R2 and NY11 had lower coefficients of determination because these data points can be considered as possible subsets of the entire range of porosities and bulk densities found at NY6. The scatter of the points along the regression line at each station is a function of the distance of that station from the dumpsite, and therefore the degree to which the

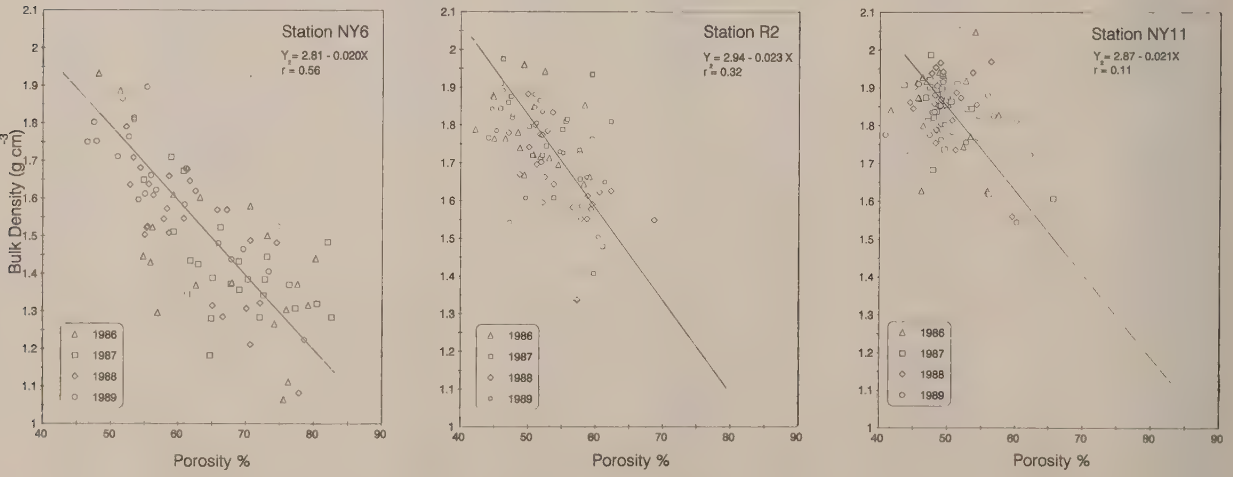

Figure 5
Geometric mean regression analyses of porosity versus saturated bulk density for stations a) NY6, b) R2, and c) NY11 in the New York Bight apex. Data points represent monthly averages.

natural sediments have been diluted by the sludge and, at NY6, the dispersion of that sludge over time.

Total Organic Carbon (TOC)

The concentration of TOC in the sediments at NY6 was highly elevated relative to R2 and NY11 (Fig. 6). Concentrations of TOC at NY6 were extremely variable (CV=62.4%) and ranged from 7.29% in December 1986 to 0.59% in September 1989; several very high and low values occurred during both pre- and postcessation periods and had no obvious seasonal pattern. Sediment TOC content decreased after the cessation of dumping as evidenced by a highly significant difference in pre/postcessation mean TOC at NY6 ($p<0.001$; Table 1). The sediment at R2 was also enriched in TOC, although to a much lesser degree and with less temporal variability (CV=54.1%). Unlike NY6, the sediment organic carbon at R2 did not decline after December 1987 (mean=0.93%). NY11 had the lowest TOC content, less than 0.6%, and the lowest variability (CV=33.8%). Mean TOC at this station showed a small but significant ($p<0.5$) increase over time, rising from an average of 0.26% before cessation to 0.33% after (Table 1). Pre- and postcessation means of NY6 relative to R2 and NY11 showed highly significant differences ($p<0.001$; Table 1); R2–NY11 differences were insignificant ($p>0.05$).

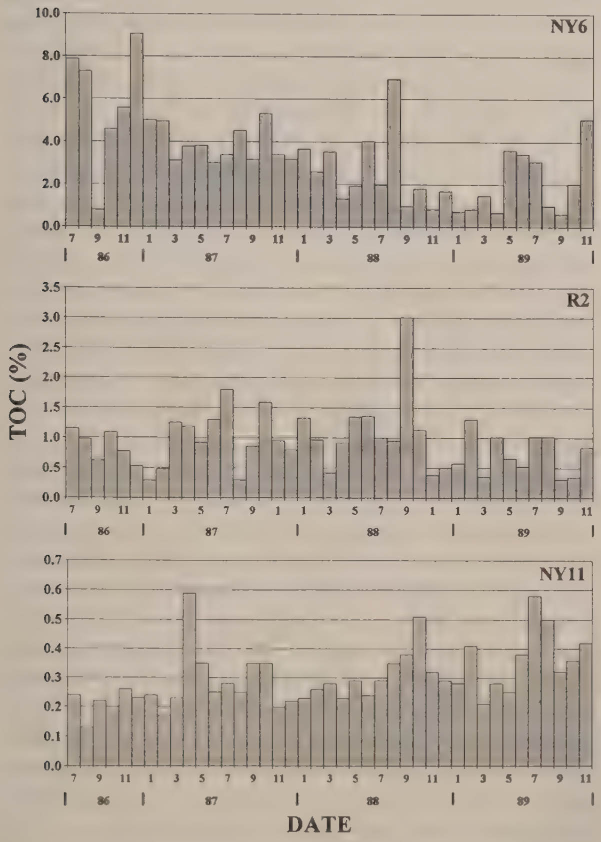

Figure 6
Sediment percent total organic carbon at 0–1 cm depth at the three replicate stations (NY6, R2, NY11) in the New York Bight apex.

Discussion

The large-scale morphologic features, natural surficial sediments, and sediment transport within the New York Bight have been previously mapped and discussed by numerous authors (Schlee, 1973; Williams and Duane, 1974; Schlee and Sanko, 1975; Freeland et al., 1976, 1979, 1981; Harris, 1976; Swift et al., 1976; Williams, 1976; Stubblefield et al., 1977; Freeland and Swift, 1978; Vincent et al., 1981; Clarke et al., 1983; and Young et al., 1985). The floor of the New York Bight shelf is predominately sand with fine-grained (2–4ϕ), poorly sorted muddy (10–40% silt-clay) sand occurring in enclosed lows of the Hudson Shelf Valley axis and the upper end of the Christiaensen Basin; sandier sediments are also found on the flanks of the Valley. The northern periphery of the Christiaensen Basin is characterized by possible cyclic expansion and contraction of mud (50% >4ϕ) facies; the nearshore Long Island mud patches are often covered with a sand cap that is subject to scouring. The remainder of the New York Bight apex contains assorted sizes of sand and both anthropogenic and natural gravel deposits.

Suspended sediment is transported or dispersed through many feedback loops operating in the inner shelf; sediments are exchanged between the Hudson estuary, the New York Bight apex, Christiaensen Basin, and the Hudson Shelf Valley (Young et al., 1985). The Shelf Valley is a "sink" for the net southwestward transport of sediment along the shelf, even though the currents and net potential sediment transport rates are stronger within the Valley than the surrounding shelf. Overall the distribution of bottom sediment types in the New York Bight apex is considered to be patchy with small scale spatial variations, although Stubblefield et al. (1977) state that the distribution of grain sizes of the sediments have a remarkable degree of stability and have no systematic seasonal variation, indicating that the seabed is in a state of textural equilibrium with the hydraulic climate. However, because transport of sediment in the Bight is dominated by storm events (Lavelle et al., 1978; Young et al., 1981; Clarke et al., 1982; Young and Hillard, 1984), several authors have noted that the Christiaensen Basin shows major temporal variability in sludge accumulation, sediment character, and mass resuspension events (Freeland et al., 1979; Vincent et al., 1981; Davis et al., 1995; Manning, 1995).

In this study, the background sediments at all three replicate stations in the Christiaensen Basin can be characterized as a poorly sorted, very fine sand throughout the study period, although a significant muddy fraction was present at both the highly polluted station, NY6, and the enriched station, R2. The highest percentages of sand were found at NY11 as it is situated on the eastern shoulder of the Hudson Shelf Valley. Davis et al. (1995) demonstrated that because sewage sludge contributed a fine component to the sediments, strong temporal fluctuations in sediment character were evident at sites adjacent to the dumpsite, such as NY6. These sediments were often characterized by a black, highly erodible, watery silt alternating with stable, homogeneous silty-sand. Thus, Davis et al. (1995) suggested that NY6 accumulated sludge during calm periods prior to the cessation of dumping and then lost it during storm events, which exposed the sandy base. This sequence of events would also explain the higher variability over time in the average grain-size and percent fine fraction we observed at NY6. Since cessation, Davis et al. (1995) have observed the gradual replacement of this anoxic silt by a muddy sand. In addition, Davis et al. (1995) found a precessation anoxic muddy sand at R2, but little evidence of sludge accumulation in the form of watery silt. This station was also subject to sediment deposition and to winnowing.

Our results showed no significant change in the mean grain-size of the inorganic fraction of the sediment at the three stations after the cessation of dumping. But owing to the nature of the overlying current regime, a geologic fine fraction is present at each of the three stations in varying amounts. NY6 had the highest mean phi (smallest diameter particles) and highest percent of fine fractions throughout the study period as compared with R2, which in turn had overall higher mean phi and percent fines than at the reference station NY11. It should be noted, however, that any inorganic material in the sediment that may have been contributed by the sludge was not distinguishable by our analyses.

Sorting is the approximation of the standard deviation in a grain-size distribution. A well-sorted sediment is one in which all the grains are of similar size; poorly-sorted sediments have a mixture of sizes. Folk (1980) states that currents of relatively constant strength will produce better-sorted sediments, whereas poorly-sorted sediments are the result of an intermittent and widely fluctuating current regime. The replicate stations contained sediments that were overwhelmingly poorly sorted throughout the study period due to erosional–depositional events observed by Davis et al. (1995) that coincide with the wind-driven currents in the Christiaensen Basin (Manning, 1995).

Skewness (or asymmetry) and kurtosis (or peakedness) indicate how closely the grain-size distribution approximates the normal Gaussian probability curve; the grain-size curve deviates from normality when these indices become extreme. Sediment distributions which are dominated by one end member with only a small amount of the other end member are extremely leptokurtic and skewed, as are the majority of our samples, and the sign of the skewness depends on which end member is dominant (Folk, 1980). A positively skewed distribution means there is an abundance of coarse grains with a surplus of fine grains in a tail, while a negative skewness shows an abundance of fine grains with a surplus of the coarser sediments in a tail. Bimodally distributed sediments such as those found at NY6 in July 1986 show extreme skewness and platykurtosis with subequal amounts of the two end members, although the pure modes of the tail ends may themselves have nearly normal distribution curves (Folk and Ward, 1957; Folk, 1980). Skewness may be the result of size-selective deposition and erosion of a sedimentary deposit (McLaren, 1981). If, as in the Christiaensen Basin, the sedimentary environment is dynamic and the winnowing action of the currents is intermittent, the skewness of the grain-size distribution curve will be affected such that neither positively nor negatively skewed sediments are dominant for any length of time (Duane, 1964) and skewness will fluctuate temporally. This pattern was seen at NY6.

Accumulation of sewage sludge prior to cessation, as well as the continuing presence of mud in the often energetic environment of NY6, indicates that there are high concentrations of suspended particulate materials from both the dumpsite and natural sources available for deposition during any relatively quiescent period. The persistence of sludge or mud at this station is dependent upon the time to the next period of intense current activity containing sufficient energy to overcome the cohesive forces of the sludge/mud deposit (Stanford et al., 1981). The sediment grain-size distributions at R2, which are almost uniformly fine-skewed and leptokurtic throughout the entire study, reflect this pattern to a lesser degree. That is, sludge input was not as important a factor (Davis et al., 1995; also seen in our TOC results) and the currents are perhaps more constant with fewer relatively calm periods which would allow mud to either settle or remain on the bottom. The sediment grain-size distributions at NY11, which are predominately near-symmetrical and also very leptokurtic, are the result of a higher energy environment, particularly as found on the flank of the Hudson Shelf Valley in the winter months where west-northwest winds produce an up-valley flow often in excess of 25 cm/s^1 for several days duration (Nelson et al., 1978; Manning, 1995). The current regime here may be relatively more constant than at the other stations, even during the remainder of the year when the magnitude of the flow is half of what it was in the winter and the direction of flow is more variable (Manning, 1995).

Porosity is a measure of the amount of pore space between sediment grains and is dependent upon the fundamental properties of size and shape distribution, as well as mineralogy, packing of the grains, and for silt-clay sediments, the concentration of organic carbon (Keller, 1982; Bennett et al., 1985; Chassefiere, 1987; Chassefiere and Monaco, 1989). Under natural sedimentary processes, sand or larger-size particles in suspension sink rapidly out of the water column and may be transported along the bottom. Under the influence of gravity or water motion, the particles assume positions in direct contact to the other grains in the seabed. These marine sands, with their grain-to-grain contacts, vary in porosity from about 35–50%, and average around 40% for medium sands and 44% for fine sands (Hamilton, 1970). However, the sediment structure formed by silt-clay particles is not controlled as much by gravity as by the adsorbed water around the grains and by interparticle forces. Suspended fine particles flocculate and form loose, open three dimensional aggregates that sink to the bottom and have a distinctly different structural framework than those of sands (Hamilton, 1970; Bennett et al., 1981). Therefore, decreasing grain-size results in a general increase in porosity. Also, as seen in this study and others, porosity is inversely related to bulk density, and this relationship is linear (Davis, 1954; Murty and Muni, 1987). If the water content of the sediment is close to zero, the saturated bulk density will be the density of the solid material, but as water content increases, the bulk density will decrease until it approaches the density of the seawater (Lee and Chough, 1987).

Since the in situ sewage sludge contaminated sediments at NY6 contain a greater silt fraction than the less contaminated sediments from the other stations (Davis et al., 1995), and because organic carbon is a contributor to high porosity and thus high water content (Chassefiere, 1987, Chassefiere and Monaco, 1989), it was expected that NY6 sediments would have high porosities and low bulk densities during dumping. After sludge dumping ceased the sediment porosity should significantly decrease and the bulk density significantly increase as this fine, organic layer was gradually winnowed, leaving a heavier inorganic fine sand fraction behind. The watery, highly organic sediments with high porosities and low bulk densities in 1986 (average of 66.88% and 1.44 g cm^{-3}, respectively) become the sandier "background" sediments with lower porosities and higher bulk densities in 1989 (average 58.60% and 1.65 g cm^{-3}, respectively) also found elsewhere in the Christaensian Basin at sites unaffected by dumping. Freeland et al. (1979) hypothesized that in the New York Bight apex, there was not only a high rate of dispersion of the sludge, but there was also rapid biological decomposition of the organic matter and this dispersion-decomposition could also account for the postcessation increase in bulk density and decrease in porosity seen in our study at NY6. The fate of sewage-derived organic carbon is most notably remineralization (Sampou and Oviatt, 1991). Studies by Smith et al. (1973) indicated that 20% of the input of sludge could be remineralized on the bottom daily, while Nedwell and Lawson (1990) demonstrated that sewage sludge at the sediment-water interface may rapidly degrade over only a four month period in shallow marine environments, particularly at dispersive sites.

Although the mass properties of porosity and bulk density are usually correlated with the fundamental property of sediment grain-size, an increase in mean phi size would not have been seen with decreasing porosity and increasing bulk density through time at NY6 because the grain-size analysis methods required the elimination of the organic component from the sediments. Since organic matter is a significant portion of the sludge-contaminated sediments (Hatcher and Keister, 1976), our inorganic grain-size results do not reflect the contribution of any sewage-derived organic fine fraction material to the sediments as do the results from our porosity and bulk density determinations because the organic matter was retained for these analyses.

In sediments, organic matter unaffected by anthropogenic inputs is derived from either soft tissues or skeletal remains. Soft tissues are composed primarily of carbonaceous compounds while skeletal remains are composed of carbonates, silica, or phosphate compounds. As carbon is the most abundant element, measurement of total organic carbon (TOC) in the sediments (percent dry weight) is a reliable way of estimating the abundance of biogenous material (Gross, 1971). Concentrations of TOC are related inversely to the mean grain-size of the sediment particles and the geotechnical property of bulk density, and TOC is related directly to the geotechnical properties of sediment porosity and both liquid and plastic limits (Keller, 1982; Jocteur Monrozier et al., 1983; Bennett et al., 1985; Booth and Dahl, 1986; Chassefiere, 1987; Mayer et al., 1988; Chassefiere and Monaco, 1989). Because the major source of the organic matter in the New York Bight apex sediments is allochthonous sludge loading (Hatcher and Keister, 1976), a decrease in sediment TOC and porosity and an increase in bulk density occurred at the polluted sites after the cessation of dumping.

Previous studies on organic matter content in New York Bight apex sediments by Gross (1970, 1972) and Reid et al. (1982) showed total carbon concentrations ranging from 0.2–4.5% in the vicinity of the sludge dumpsite to <0.2% for sediments more than 10 km from waste disposal activities (as at NY11). Hatcher and Keister (1976) reported TOC values in the Bight apex and adjacent Hudson Shelf Valley in 1973 to be <0.1% dry wt in sands to around 5% in deeper muds. Harris (1976) also reported TOC values of 0.1–4.5% for the dumpsite muds and Christiaensen Basin. Results from our study demonstrate that TOC in the sediments of NY6 showed a significant response to the cessation of dumping, decreasing from precessation mean levels of 4–5% to levels of around 2% following cessation. Lower but still enriched mean TOC values (0.9%) were observed at R2 throughout the study period, while NY11 maintained mean TOC concentrations similar to the background levels observed in the studies mentioned above. The elevated mean TOC levels at NY6 and R2 at the end of the study, along with the elevated porosity values and depressed bulk density values at these two stations as compared with NY11, are simply a reflection of the greater percentages of silts and clays still present in these sediments. However, at this time there is no explanation for the slight postcessation rise in TOC at NY11. Finally, the TOC time series data showed evidence of temporal variability in the order of NY6>R2>NY11, matching the order we observed for grain-size, porosity and bulk density.

In this paper, we have sought to characterize the dumpsite sediment milieu and detect the subtle patterns and interrelationships among the sediment variables. This has allowed us to draw conclusions regarding correlations between the sediment properties and the cessation of sludge dumping. These conclusions will assist us in the future interpretation of the trends observed in the associated dumpsite chemical and biological data.

Summary

This study described the fundamental and derived properties of the surficial sediments from three sites in the vicinity of the 12-mile dumpsite in the New York Bight apex during the period 1986 through 1989. The RIT or BACI design of Pikanowski (1992, 1995) and Stewart-Oaten et al. (1986) was used to test for significant changes in grain-size characteristics, porosity, bulk density and total organic carbon related to the cessation of sludge dumping in December 1987. Surface sediment samples were taken before and after the cessation of sludge dumping at a reference station, NY11, an "enriched" station, R2, and the degraded station, NY6, located near the dumpsite. The sediments at all three sampling stations were poorly sorted, very fine sand (means of 3.1–3.5ϕ) although NY6 and R2 contained greater mud fractions than NY11. Skewness of the grain-size distributions over the 41 month study period at NY6 ranged from very coarse to very fine-skewed while those at R2 were almost uniformly fine-skewed and NY11's distributions were predominately near-symmetrical. Nearly all the grain-size distributions were leptokurtic. The poorly sorted sediments and skewed (especially at NY6 and R2) and leptokurtic grain-size distributions are an indication of intermittent and widely fluctuating bottom currents and sediment resuspension events. Previous and concurrent studies of sediment transport in this area support this conclusion. NY6 had the greatest response to local erosional–depositional events by having a higher variability in mean grain-size diameter, percent fine fraction, inclusive graphic skewness, porosity, saturated bulk density, and percent TOC throughout the study period. These same sediment properties at NY11 generally had the lowest temporal variability, while values at R2 were generally intermediate between NY11 and the heavily degraded NY6. There was no significant change in the mean grain-size of the inorganic fraction of the sediment at the three stations after the cessation of dumping. Although the required digestion of the organic matter during our grain-size analyses removed the organic fine fraction component attributable to the sludge, the organic material was retained in the sediment samples analyzed for bulk density, porosity and TOC. Thus, the results from these analyses at the degraded station NY6 showed a significant response to the cessation of sewage sludge dumping. Since the in situ sewage sludge contaminated sediments at NY6 contained a greater silt fraction than the less contaminated sediments from the other two stations, these NY6 sediments had higher porosities, lower bulk densities (these variables are additive inverses) and higher percent TOC prior to the cessation of dumping in 1987. NY6 sediment porosity significantly decreased from mean precessation levels of 68% to postcessation levels of 61%, while TOC levels decreased from a precessation mean of 4–5% to around 2% following the cessation of dumping (porosity and TOC are directly related). NY6 bulk density significantly increased from a precessation mean of 1.4 g cm^{-3} to 1.6 g cm^{-3} after dumping ceased. These postcessation changes are due to rapid mineralization of the organic matter and to the dispersion of the fine sludge layer, leaving a heavier sand fraction behind. Average percent TOC at R2 (0.9%) was consistently higher than at NY11 (0.3%). In general, R2 displayed intermediate values between those of NY6 and NY11 for all variables tested. The higher TOC and porosity values and lower bulk densities found at NY6 and R2 at the end of the study as compared to NY11 are a function of the higher percentages of silts and clays still present in these stations' sediments.

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Introduction

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Almost two decades have passed since 1972 when the Environmental Protection Agency (EPA) was required under the Ocean Dumping Act to consider a number of factors in establishing regulations for ocean disposal of wastes. Chief among them were effects on health, fisheries, and marine resources. In 1984, during suits brought by New York and New Jersey to continue use of the site, a number of biological effects, attributed wholly or in part to sludge dumping, were listed as contributing to EPA's decision not to redesignate the 12-mile site. Included among these were:

1) Reduced catches of bony fishes;

2) Impaired health and well-being of resource species, including reduction of reproductive functions, increased incidence of disease, e.g. fin rot and black gill, and mutation of fish larvae;

3) Alterations in the benthic community, particularly the proliferation of pollution-tolerant polychaete worms such as *Capitella*; and perhaps most importantly,

4) Unacceptably high levels of coliform bacteria which forced closure of some 96,000 acres of shellfish beds by FDA. This threat to public health was, of course, one of the most significant factors in EPA's decision.

Consequently, any study designed to evaluate the effects of closure of the dumpsite, had to include a major component to measure changes on the biota.

The choice of biotic variables was based on relevance to fisheries of the inner Bight in terms of abundance, distribution, and contamination of resource species, and on predictive value for detecting changes with abatement. Similar to the approach used for the physical and chemical portions of the study, working hypotheses were developed based on anticipated changes in a number of biotic variables. Each of these hypotheses was formulated from the study's principle null hypothesis that, for effects that are reversible, the difference in ecology between an affected site and an unaffected site does not change after the cessation of sewage sludge disposal (EPD [Environmental Processes Division], 1988).

Returning to EPA's testimony that sludge dumping contributed, in part, to reduced catch of fish, the most obvious objective was to determine the distribution and abundance of demersal finfish and the larger megainvertebrates, such as crabs and lobsters, which were dependent on habitats in and around the dumpsite. While a major survey was needed to identify the extent to which this area was utilized by a broad range of species present in the New York Bight, it was also deemed important to focus on a few target organisms.

Winter flounder, *Pleuronectes americanus*, is one of the more commercially and recreationally important demersal species in the Bight. However, little was known about the extent to which flounder from coastal New Jersey migrated to offshore habitats influenced by sewage sludge. If a portion of this population was dependent on habitats affected by dumping at the 12-mile site, then possible links could begin to be established for pollutant effects, i.e. contaminant bioaccumulation or disease. Thus, we examined the premise that winter flounder that utilize the areas in and around the dumpsite comprised a significant portion of the population of Sandy Hook and Raritan bays.

One of the biological effects noted by the EPA was the impairment of resource health. Surveys in the New York Bight in the early to mid–1970's identified a number of diseases of marine organisms which could have been indicative of poor water quality. Given the anticipated decrease in levels of contaminants in the dumpsite area, would there be a corresponding decrease in the incidence of disease? The study focused on gross pathological indicators, e.g. fin erosion, ulceration, parasitism, tumors and skeletal abnormalities, anticipating that incidence in winter flounder would significantly decrease following cessation of dumping.

Another signal of biological response could be expected to be found in changes in trophic patterns. Even if there were no signficant changes in distribution and abundance of finfish and megainvertebrates, reductions in contaminant input could result in major shifts in diet and food habits. Food habits were examined for key resource species of the New York Bight, e.g. red hake, *Urophycis chuss*; lobster, *Homarus americanus*; and winter flounder. The working hypothesis was that the diets of these species would change following shifts in the availability of benthic prey at the replicate sites.

Changes in numbers and species of benthic organisms have long been accepted as indicators of biological effects in contaminated environments. Based on numerous surveys over the years, there is an extensive data base on benthic macrofauna of the New York Bight that can serve as a baseline against which to measure change. Benthic studies have also been conducted in other areas in which sludge dumping has occurred including England, Scotland, and Germany as well as the Philadelphia site (see Reid et al., 1995, for review). Since changes in benthic community structure could also be correlated with possible changes in food habits of demersal predators, an understanding of the effects of cessation of dumping on benthic macrofaunal diversity and species richness was a critical component of the present study.

Using expected patterns of benthic response to reduction in contaminants, several working hypotheses were developed: first, significant increases would occur in the numbers of benthic species and crustacean forage organisms at the enriched station R2 and even more probably at the highly degraded NY6; and second, a significant reduction would occur at NY6 in the areal coverage and mean density of *Capitella*, a polychaete identified with polluted substrates.

The most pressing reason for closure of the dumpsite was concern for public health stemming from unacceptably high levels of coliform bacteria in shellfish harvesting areas. In 1983, a technical report prepared by New York City's Department of Environmental Protection[1] to support redesignation of the 12-mile dumpsite stated that based on coliforms, sewage sludge contributed only 0.2% of the microbial loading to the Bight, with close to 99% attributable to the Hudson–Raritan plume. It was estimated, therefore, that if dumping of sewage sludge were the only source of fecal contamination in the Bight, it was likely that the shellfish beds could be reopened within a year after cessation. However, surveys conducted before and after closure of the more dispersive Philadelphia dumpsite, indicated that it took nearly four years after closure before microbial contamination returned to background levels (Devine and Simpson, 1985).

The study would not have been complete therefore, without an evaluation as to whether, after 64 years of sludge input, bacterial contamination would decrease to acceptable levels and permit the shellfish beds to be reopened.

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Response of Fish and Megainvertebrates of the New York Bight Apex to the Abatement of Sewage Sludge Dumping—an Overview

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ABSTRACT

The National Oceanic and Atmospheric Administration, National Marine Fisheries Service, conducted a 39-month study, from July 1986 to September 1989, to assess effects of the end of sewage sludge dumping on the living marine resources and their ecological relationships in the New York Bight apex. During the study 991 otter trawls were made at 19 to 25 stations per collecting interval. Abundance and composition were enumerated for 75 species, representing 46 families of fish and megainvertebrates. The New York Bight apex hosted a wide variety of migratory species; however, the demersal biomass was dominated seasonally by only a few. Atlantic rock crab, *Cancer irroratus*, was the most abundant in summer, replaced by little skate, *Raja erinacea*, and spiny dogfish, *Squalus acanthias*, during winter.

Five species made up >75% of the fish biomass: little skate; spiny dogfish; winter flounder, *Pleuronectes americanus*; red hake, *Urophycis chuss*; and ocean pout, *Macrozoarces americanus*; little skate and winter flounder were obtained at >80% of all stations. Four species made up >90% of the megainvertebrate biomass: Atlantic rock crab; horseshoe crab, *Limulus polyphemus*; American lobster, *Homarus americanus*; and longfin squid, *Loligo pealeii*; Atlantic rock crab occurred at >85% of all stations.

The null hypothesis of no difference in biomass due to abatement of sludge disposal was tested for total fish and megainvertebrates as well as for the most frequently occurring species. Total fish and megainvertebrates, Atlantic rock crab, little skate, and winter flounder biomass did not change significantly after cessation of dumping. Only American lobster biomass changed significantly, increasing after dumping stopped.

Introduction

The New York Bight (NYB) is that portion of the western North Atlantic bounded by the coasts of Long Island, New York to the north, New Jersey on the west, and extending seaward to the edge of the continental shelf (Fig. 1). This study was conducted in the NYB apex, i.e. the area where the Long Island and New Jersey coastlines are nearly perpendicular and the ocean is contiguous with the Hudson–Raritan estuary (Fig. 1). This portion of the NYB is directly adjacent to one of the most heavily industrialized and densely populated regions in the United States and, consequently, is vulnerable to all the problems associated with multiple uses of finite resources (Mayer, 1982; Squires, 1983). The waters of the NYB, and the apex in particular, are used heavily for transportation, commercial and recreational fishing, pleasure boating, and waste disposal from the dense population of ≈20 million people who live, work, and recreate on its edges (Gross, 1972, 1976; Pararas-Carayannis, 1973; U.S. Environmental Protection Agency, 1975; Wilk and Baker, 1989).

Sewage sludge was dumped at a site 22.2 km (12.0 nmi) from the tip of Sandy Hook (Fig. 1) since 1924 (Swanson et al., 1985; Santoro, 1987). Between March 1986 and December 1987, disposal of sludge was phased out at the 12-mile dumpsite (12-MDS) (Fig. 2) and redirected to a dumpsite 196 km (106 nmi) offshore.

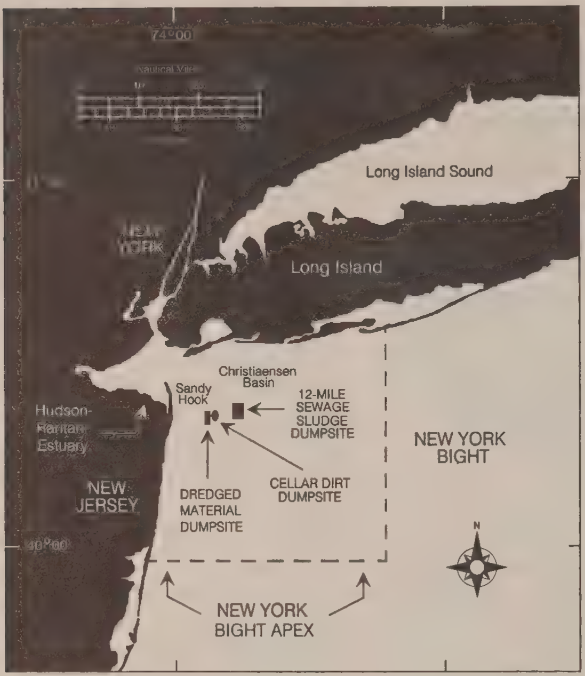
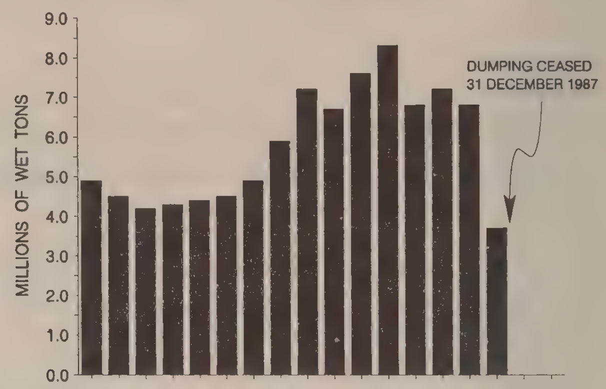


Figure 1

New York Bight (NYB) segment of the Middle Atlantic continental shelf including outline of NYB apex as well as the locations of various designated dumpsites within the apex.



YEAR	72	73	74	75	76	77	78	79	80	81	82	83	84	85	86	87	88	89
WEIGHT	4.9	4.5	4.2	4.3	4.4	4.5	4.9	5.9	7.2	6.7	7.6	8.3	6.8	7.2	6.8	3.7	0.0	0.0

Figure 2

Estimated annual amounts of sewage sludge dumped in the New York Bight apex, 1972–89 (Swanson et al. 1985; Santoro 1987).

The NYB apex also receives contaminant inputs from other sources besides sewage sludge; the two most significant being the Hudson–Raritan plume and the dredged material dumpsite (Fig. 1) (New York City Department of Environmental Protection[1]). This multiplicity of sources has made it virtually impossible to quantify and separate contamination and biological effects due directly to the dumping of sewage sludge. In addition, the lack of a pre-dumping baseline at the 12-MDS further hinders understanding of the effects of sludge. Among the few existing pre- and post-dumping data are results from surveys conducted by NOAA, U.S. Environmental Protection Agency (USEPA), and the Food and Drug Administration before and after the 1980 closure of the Philadelphia sewage sludge dumpsite (Devine and Simpson, 1985). That study indicated that about four years were needed for microbial contamination to return to what were considered "background levels."

Periodic resource surveys and characterizations of the fish and fisheries of the NYB, which describe the distribution, abundance, community structure, and aspects of the life history of selected species, have been conducted during the past three decades (Wilk et al., 1977, 1983a, 1983b, 1988; Wilk and Baker, 1989). In addition, other studies have tried to characterize the effects of anthropogenic inputs on the sediment and benthos of the NYB (Gross, 1976; Mayer, 1982; Reid et al., 1987). All these studies had inherent drawbacks which included the inability to: 1) separate the effects of dumping sewage sludge from effects due to dredged and riverborne materials; 2) substantiate that "control" stations were significantly different from "contaminated" stations; 3) account for reducing conditions in the sediment, which occur episodically during the summer in organically contaminated areas of the NYB, that could markedly influence community structure (EPD [Environmental Processes Division], 1988); and 4) improve the accuracy and precision of individual studies, because environmental and biological variables were not measured systematically or synoptically at appropriate sampling intervals.

The abatement of dumping sewage sludge at the 12-MDS offered an opportunity to determine ecosystem response to the removal of a major anthropogenic input. NOAA's National Marine Fisheries Service (NMFS) conducted a multidisciplinary study to determine if cessation of dumping at the 12-MDS would result in significant changes in the abundance and species composition of fish and megainvertebrates, water and sediment chemistry, benthic community structure, and body burdens of organic contaminants, and would lead to reopening of shellfish beds (EPD, 1988).

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[1] New York City Department of Environmental Protection. 1983. Technical information to support the redesignation of the 12-mile site for the ocean disposal of municipal sewage sludge. Prepared by Ecological Analysts, Inc. Sparks, MD, and SEAMOcean, Inc., Wheaton, MD. New York City Department of Environmental Protection, 2358 Municipal Bldg., New York, NY 10007. Unpubl. manuscr. 483 p. + appendices.

This paper summarizes the following aspects of the study: 1) the abundance and species composition of the fish and megainvertebrates collected; and 2) the tests of the null hypothesis of no effect attributable to the abatement of sewage sludge disposal upon the abundance of all finfish and megainvertebrates as well as of several common species.

Methods and Materials

Survey Design

Before the phaseout of the 12-MDS, it was thought possible to provide a better understanding of the effects of dumping sewage sludge on habitats and biota and the interactions and relationships among them by conducting synoptic surveys (EPD, 1988). Ideally, replicate measurements should be collected at all sample locations so contaminated and adjacent "cleaner" sites could be characterized sufficiently over time. Replicated sampling at all sites, however, was judged too costly and labor-intensive. Therefore, the survey design was a compromise consisting of two complementary sampling regimens that are termed replicate and broadscale surveys (see Pikanowski, 1995).

The replicate survey design assumed that changes in sediment chemistry, which may affect fish and megainvertebrates, occur on the order of months. Therefore, sampling to detect these changes and possible effects must be made frequently, especially during summer when stressful combinations of high temperature, low dissolved oxygen (hypoxia), and high sulfide are most likely to occur (EPD, 1988). Replicate sampling was conducted at three stations chosen to represent a gradient of sewage sludge concentrations and effects (Fig. 3). The broadscale survey design consisted of a single, non-replicated suite of measurements taken at 19 to 25 stations that covered much of the NYB apex (Fig. 3). Stations were selected to include all major habitat types that would be affected by change in sludge input. Details of the sampling design can be found in EPD (1988) and Pikanowski (1992).

Station Selection

Replicate stations were selected based on the similarity of depth and sediment type, the presumption of some isolation from dredge material dumping, and the need to achieve the broadest range of influence with respect to sewage sludge, i.e. Station NY6="polluted", Station R2="enriched", and Station NY11="control" (Fig. 3; EPD, 1988). Station locations for broadscale surveys were selected to: 1) relate this effort with the replicate survey, i.e. 19 to 25 stations per survey; 2) provide coverage of the area of apparent influence of the sewage sludge dumping to facilitate mapping and contouring; 3) provide complete coverage of the ecological gradients in the study area to legitimize correlative analyses; and 4) correspond with historical stations whenever possible (EPD, 1988).

Sampling Schedule

Replicate and broadscale surveys were conducted year-round during alternate months. Replicate sampling was done monthly from July to September, when hypoxia and sulfide accumulation were most likely to occur; consequently, both replicate and broadscale surveys were conducted each August (EPD, 1988). Sampling began with a replicate survey in July 1986 and continued through September 1989, 21 months after the end of the dumping of sewage sludge in December 1987.

Scheduling within a replicate survey was complex. A balanced sampling design was devised to control potential photoperiod bias by alternating the station sampling order at each replicate station (Table 1). Otter trawls, grab samples, and bottom water samples were replicated eight times at each station. Three sets of

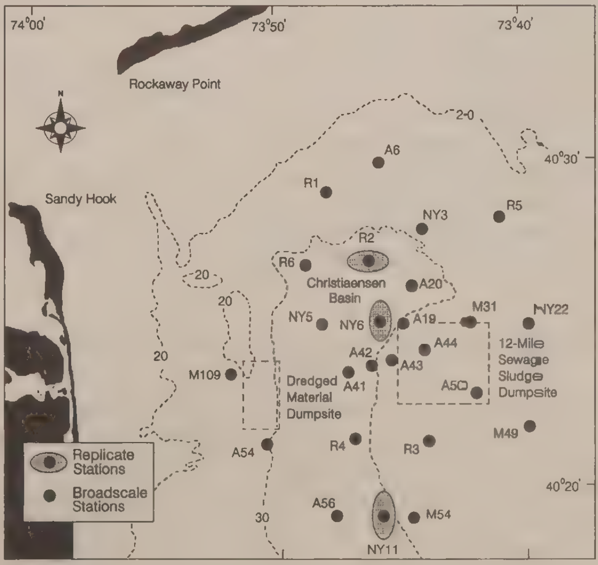


Figure 3

Location of trawl stations sampled during a fish-megainvertebrate survey conducted in and around the 12-mile sewage dumpsite, July 1986–September 1989. Isobaths are given in meters.

Table 1

Station sequence employed during the replicate portion of a trawl survey conducted in and around the 12-mile sewage dumpsite, July 1986 - September 1989. R-Tow=tow radially through site center; D-Tow=tow distal from site center (EPD, 1988).

Sampling Day	Photoperiod	
	AM	PM
1	NY11 2 R-Tows	NY6 1 R-Tow 1 D-Tow
2	NY6 2 R-Tows	R2 1 R-Tow 1 D-Tow
3	R2 2 R-Tows	NY11 1 R-Tow 1 D-Tow
4	NY11 2 D-Tows	R2 2 D-Tows
5	NY6 2 D-Tows	NY11 2 D-Tows
6	R2 2 D-Tows	NY6 2 D-Tows

measurements were made at the central point; the remaining five were taken within an ellipse about each point, with the major axis of each ellipse oriented to cover the same depth.

The sequence of sampling in the broadscale surveys was straightforward; a single trawl tow, bottom water sample, and grab sample were taken at each station. The number of stations and order of sampling depended on weather and vessel scheduling. Additional details of station selection and sampling schedule are given in EPD (1988) and Pikanowski (1992).

Collection of Fish and Megainvertebrates

Fish and megainvertebrates were collected with an otter trawl having an 11.0-m (36-ft) footrope and a 9.8-m (32-ft) headrope. The body and cod end were constructed of 76-mm (3-inch) and 51-mm (2-inch) stretch mesh nylon, respectively. The trawl was towed for 15 min at approximately 4.0 km/h (2.4 kn) with tows made along isobaths to avoid sudden depth changes. Although tow time was kept constant, direction and distance of each tow were affected by current, tide, wind, and, in some cases, the need to shift heading to avoid lobster pots.

Catches were emptied on deck and all fish and megainvertebrates were identified, each taxa weighed to the nearest 0.1 kg, and individuals measured to the nearest 1.0 cm. All specimens were measured except when large catches required subsampling. In such cases, an expansion factor (weight of total catch/weight of subsample) was applied to estimate the number in the total catch. Additional information about the collection methodology can be found in Wilk and Baker (1989) and Wilk et al. (1992).

Results and Conclusions

Catch Analysis

In 39 months during 43 cruises, 991 trawl stations were made. Table 2 lists cruise codes, dates, number of stations, and survey type for the study. Figure 4 illustrates seasonal abundance of mean total as well as fish and megainvertebrate catch (kg/tow) over the course of the survey. A further breakdown of total catch into replicate and broadscale segments, including the contribution of fish and megainvertebrate components, is given in Figure 5.

Seventy-five species representing 46 families of fish and megainvertebrates were captured (Table 3). As expected, since otter trawling was the method of collection, demersal finfish and large crustaceans dominated the catch (Fig. 6). Fifty-eight fish species, representing 32 families, totaling 65,255 individuals and weighing 14,113 kg, were collected during the study. Eleven species (spiny dogfish, *Squalus acanthias*; little skate, *Raja erinacea*; silver hake, *Merluccius bilinearis*; red hake, *Urophycis chuss*; ocean pout, *Macrozoarces americanus*; scup, *Stenotomus chrysops*; cunner, *Tautogolabrus adspersus*; butterfish, *Peprilus triacanthus*; fourspot flounder, *Paralichthys oblongus*; windowpane, *Scophthalmus aquosus*; and winter flounder, *Pleuronectes americanus*) comprised 90% of the fish biomass (Fig. 7A). Seventeen species of megainvertebrates, representing 14 families, totaling 48,608 individuals and weighing 4,571 kg, were collected during the study. Eight species (longfin squid, *Loligo pealeii*; northern shortfin squid, *Illex illecebrosus*; horseshoe crab, *Limulus polyphemus*; American lobster, *Homarus americanus*; Jonah crab; *Cancer borealis*; Atlantic rock crab, *Cancer irroratus*; lady crab, *Ovalipes ocellatus*; and starfish sp., *Asterias* sp.) comprised 99% of the megainvertebrate biomass (Fig. 7B).

The 19 species of fish and megainvertebrates listed above represented 93% of the total biomass. Seven fish and four megainvertebrate species were observed in more than 40% of the trawl samples; these were little skate (89%), Atlantic rock crab (87%), winter flounder (83%), starfish sp. (69%), silver hake (59%), longfin

Table 2

Summary of collecting intervals sampled during a trawl survey conducted in and around the 12-Mile sewage dumpsite, July 1986 - September 1989.

Cruise Code	Dates	Number of Stations	Type of Survey
	1986		
KA8607	Jul 16–18, 22–24	24	Replicate
KG8608	Aug 6–8, 12, 13, 15	24	Broadscale
KA8608	Aug 20, 26, 29; Sep 2, 5, 8	24	Replicate
KA8609	Sep 22–26, 29	24	Replicate
KG8610	Oct 15–17, 20–22	23	Broadscale
KA8611	Nov 14, 17, 18, 24–26, 28	24	Replicate
KG8612	Dec 11, 12, 15, 16, 22	22	Broadscale
	1987		
KA8701	Jan 14, 15, 21, 28, 30; Feb 2, 3	24	Replicate
KG8702	Feb 18–20, 25–26	22	Broadscale
KA8703	Mar 19, 20, 23–26	24	Replicate
KG8704	Apr 13–15, 20–22, 27	22	Broadscale
KA8705	May 12, 14, 15, 18, 22, 26	24	Replicate
KG8706	Jun 9–11, 15–17	22	Broadscale
KA8707	Jul 7, 13–17, 20	24	Replicate
KG/KA8708	Aug 10–14, 18–20	44	Combined
KA8709	Sep 14–18, 21, 22	26	Replicate
KG8710	Oct 16, 19–21, 26, 27	22	Broadscale
KA8711	Nov 13, 15–20	24	Replicate
KG8712	Dec 7–11, 14, 15	22	Broadscale
	12-Mile Dumpsite Closed (31 Dec 1987)		
	1988		
KA8801	Jan 11–13, 15, 19, 21	24	Replicate
KG8802	Feb 8–11, 17, 18	22	Broadscale
KA8803	Mar 7–11, 14, 15	24	Replicate
KG8804	Apr 11, 15, 19–22, 26	22	Broadscale
KA8805	May 9, 10, 12, 18, 20, 23, 24	24	Replicate
KG8806	Jun 6–8, 10	22	Broadscale
KA8807	Jul 11–15, 18, 19	24	Replicate
KG8808	Aug 8–12	19	Broadscale
KA8808	Aug 15–19, 22	24	Replicate
KA8809	Sep 12, 14–16, 19, 21	24	Replicate
KG8810	Oct 11, 12, 14, 17–20	22	Broadscale
KA8811	Nov 7–9, 14, 15, 18	24	Replicate
KG8812	Dec 6–9, 19	22	Broadscale
	1989		
KA8901	Jan 10, 11, 17–20	24	Replicate
KG8902	Feb 6, 7, 10, 13, 16	22	Broadscale
KA8903	Mar 13–17, 20	24	Replicate
KG8904	Apr 13, 14, 17–20	22	Broadscale
KA8905	May 9, 12, 18, 19, 22, 23	24	Replicate
KG8906	Jun 12–14, 16, 19, 20	22	Broadscale
KA8907	Jul 11, 12, 14, 18–20	24	Replicate
KG8908	Aug 7–10	22	Broadscale
KA8908	Aug 15, 17, 18, 21–23	24	Replicate
KA8909	Sep 11–15, 18, 20	24	Replicate

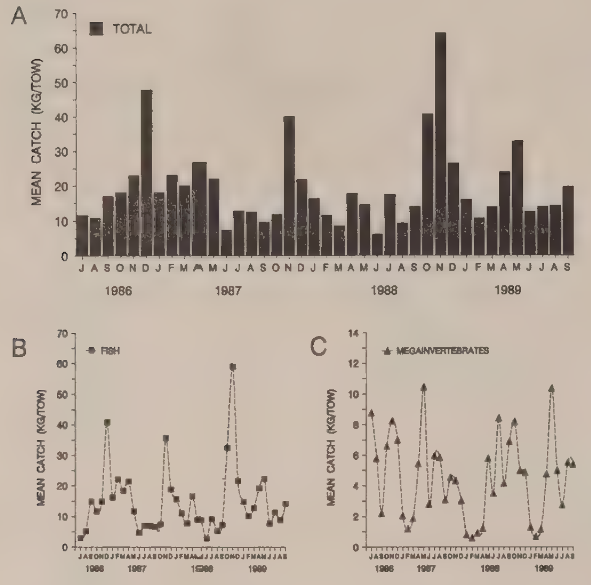


Figure 4

Mean catch (kg/tow) of fish and megainvertebrates collected during a trawl survey in and around the 12-mile sewage dumpsite, July 1986–September 1989: A) total, B) fish, and C) megainvertebrates.

squid (55%), butterfish (50%), fourspot flounder (46%), American lobster (46%), red hake (43%), and windowpane (42%). Detailed tabulations and summaries of species distribution, abundance, composition, and length range of the fish and megainvertebrates collected during the study can be found in Wilk et al. (1992).

The NYB apex hosts a diverse assemblage of fish and megainvertebrates (Table 3). However, the total biomass is dominated by a few seasonally abundant species, i.e. Atlantic rock crab in the summer and little skate and spiny dogfish during the winter. It should be noted that most, if not all, species collected exhibited some degree of seasonality. Five species (little skate, spiny dogfish, winter flounder, red hake, and ocean pout) made up >75% of the fish biomass (Fig. 7A), with little skate and winter flounder collected at >80% of all trawl stations. Four species (Atlantic rock crab, horseshoe crab, American lobster, and longfin squid) made up >90% of the megainvertebrate biomass (Fig. 7B), with Atlantic rock crab collected at >85% of all trawl stations.

Hypothesis Testing

The null hypothesis attributing no effect following the abatement of sewage sludge disposal was tested for total

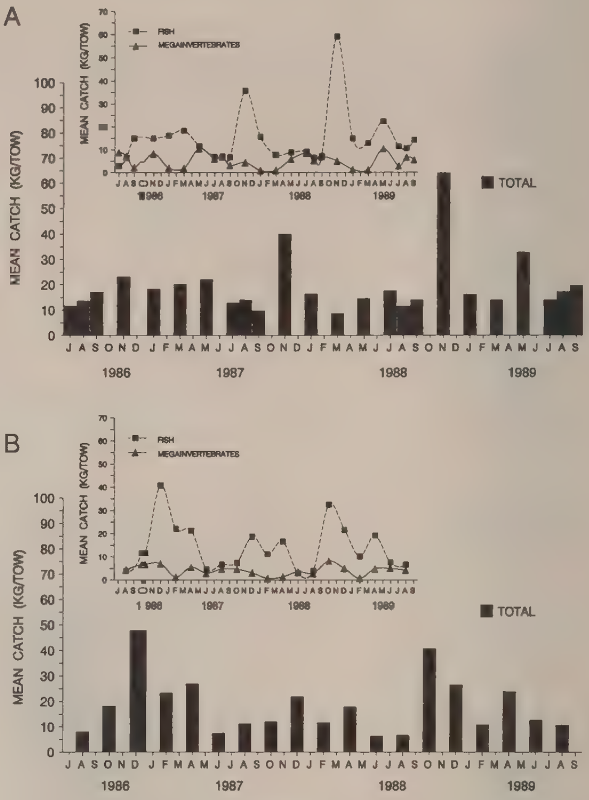
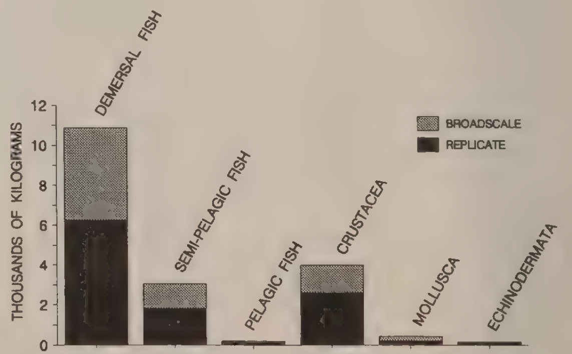


Figure 5

Mean catch (kg/tow) of fish and megainvertebrates collected during a trawl survey in and around the 12-mile sewage dumpsite, July 1986–September 1989: A) replicate surveys and B) broadscale surveys.



REPLICATE	6.249	1.821	0.134	2.607	0.209	0.105
BROADSCALE	4.619	1.225	0.065	1.383	0.213	0.054
TOTAL CATCH	10.868	3.046	0.199	3.990	0.422	0.159

THOUSANDS OF KILOGRAMS PER CATEGORY

Figure 6

Contribution, based on total biomass, by categories of fish and megainvertebrates collected during a trawl survey in and around the 12-mile sewage dumpsite, July 1986–September 1989.

Table 3

Phylogenetic listing of fish and megainvertebrates collected in and around the 12-mile sewage dumpsite, July 1986–September 1989. Fish are arranged according to Robins et al. (1991) and megainvertebrates according to Gosner (1978), Turgeon et al. (1988), and Williams et al. (1988).

FAMILY	Common Name	*Scientific Name*
	FISH	
CARCHARHINIDAE	Smooth dogfish	*Mustelus canis*
SQUALIDAE	Spiny dogfish	*Squalus acanthias*
RAJIDAE	Clearnose skate	*Raja eglanteria*
	Little skate	*Raja erinacea*
	Winter skate	*Raja ocellata*
ACIPENSERIDAE	Atlantic sturgeon	*Acipenser oxyrhynchus*
ANGUILLIDAE	American eel	*Anguilla rostrata*
CONGRIDAE	Conger eel	*Conger oceanicus*
CLUPEIDAE	Blueback herring	*Alosa aestivalis*
	Hickory shad	*Alosa mediocris*
	Alewife	*Alosa pseudoharengus*
	American shad	*Alosa sapidissima*
	Atlantic menhaden	*Brevoortia tyrannus*
	Atlantic herring	*Clupea harengus*
	Round herring	*Etrumeus teres*
ENGRAULIDAE	Striped anchovy	*Anchoa hepsetus*
GADIDAE	Atlantic cod	*Gadus morhua*
	Haddock	*Melanogrammus aeglefinus*
	Silver hake	*Merluccius bilinearis*
	Pollock	*Pollachius virens*
	Red hake	*Urophycis chuss*
	Spotted hake	*Urophycis regia*
OPHIDIIDAE	Cusk-eel sp.	*Ophidion* sp.
LOPHIIDAE	Goosefish	*Lophius americanus*
ATHERINIDAE	Atlantic silverside	*Menidia menidia*
SYNGNATHIDAE	Lined seahorse	*Hippocampus erectus*
	Northern pipefish	*Syngnathus fuscus*
TRIGLIDAE	Northern searobin	*Prionotus carolinus*
	Striped searobin	*Prionotus evolans*
COTTIDAE	Sea raven	*Hemitripterus americanus*
	Longhorn sculpin	*Myoxocephalus octodecemspinosus*
CYCLOPTERIDAE	Inquiline snailfish	*Liparis inquilinus*
SERRANIDAE	Black sea bass	*Centropristis striata*
PRIACANTHIDAE	Bigeye	*Priacanthus arenatus*
POMATOMIDAE	Bluefish	*Pomatomus saltatrix*
CARANGIDAE	Blue runner	*Caranx crysos*
	Rough scad	*Trachurus lathami*
SPARIDAE	Scup	*Stenotomus chrysops*
SCIAENIDAE	Weakfish	*Cynoscion regalis*
	Northern kingfish	*Menticirrhus saxatilis*
LABRIDAE	Tautog	*Tautoga onitis*
LABRIDAE	Cunner	*Tautogolabrus adspersus*
ZOARCIDAE	Ocean pout	*Macrozoarces americanus*
PHOLIDAE	Rock gunnel	*Pholis gunnellus*
AMMODYTIDAE	Northern sand lance	*Ammodytes dubius*
SCOMBRIDAE	Chub mackerel	*Scomber japonicus*
	Atlantic mackerel	*Scomber scombrus*
STROMATEIDAE	Butterfish	*Peprilus triacanthus*
BOTHIDAE	Gulf stream flounder	*Citharichthys arctifrons*
	Smallmouth flounder	*Etropus microstomus*
	Summer flounder	*Paralichthys dentatus*
	Fourspot flounder	*Paralichthys oblongus*
	Windowpane	*Scophthalmus aquosus*
PLEURONECTIDAE	Witch flounder	*Glyptocephalus cynoglossus*
	Yellowtail flounder	*Pleuronectes ferrugineus*
	Winter flounder	*Pleuronectes americanus*
BALISTIDAE	Planehead filefish	*Monacanthus hispidus*
TETRAODONTIDAE	Northern puffer	*Sphoeroides maculatus*
	MEGAINVERTEBRATES	
PECTINIDAE	Sea scallop	*Placopecten magelanicus*
DREISSENIDAE	Ocean quahog	*Arctica islandica*
VENERIDAE	Northern quahog	*Mercenaria mercenaria*
	False quahog	*Pitar morrhuanus*
NATICIDAE	Moon snail sp.	
LOLIGINIDAE	Longfin squid	*Loligo pealeii*
OMMASTREPHIDAE	Northern shortfin squid	*Illex illecebrosus*
LIMULIDAE	Horseshoe crab	*Limulus polyphemus*
PANDALIDAE	Bristled longbeak	*Dichelopandalus leptocerus*
NEPHROPSIDAE	American lobster	*Homarus americanus*
PAGURIDAE	Hermit crab sp.	*Pagurus* sp.
MAJIDAE	Spider crab sp.	*Libinia* sp.
CANCRIDAE	Jonah crab	*Cancer borealis*
	Atlantic rock crab	*Cancer irroratus*
PORTUNIDAE	Blue crab	*Callinectes sapidus*
	Lady crab	*Ovalipes ocellatus*
ASTERIIDAE	Starfish	*Asterias* sp.

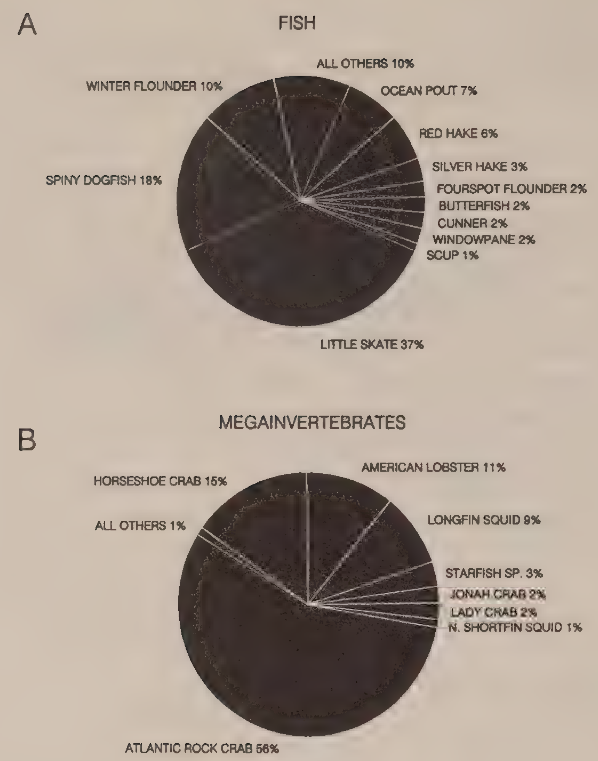


Figure 7

Percent contribution, based on total biomass, of fish and megainvertebrates collected during a trawl survey in and around the 12-mile sewage dumpsite, July 1986–September 1989: A) fish and B) megainvertebrates. See Table 3 for scientific names.

fish and megainvertebrates as well as for the most frequently occurring species (Pikanowski, 1992). The relative abundance (average weight per tow) of total fish (Fig. 8A) and megainvertebrates (Fig. 8B) as well as little skate (Fig. 8C), winter flounder (Fig. 8D), and rock crab (Fig. 8E) did not significantly ($P>0.05$) change following cessation of dumping. Of the four dominant species tested, only American lobster (Fig. 8F) significantly ($P<0.05$) increased in relative abundance after dumping stopped. This may have resulted from a redirection of commercial fishing effort, that is, lobster pots were set out in the area of the dumpsite after dumping stopped because pots were no longer in danger of either becoming fouled with or buried in sewage sludge. Based on these results, except for American lobster, the null hypothesis cannot be rejected when comparing the replicate station which received direct sludge inputs (Station NY6="polluted") with the station that received indirect sludge inputs (Station R2= "enriched") and the station that received no sludge inputs (Station NY11="control").

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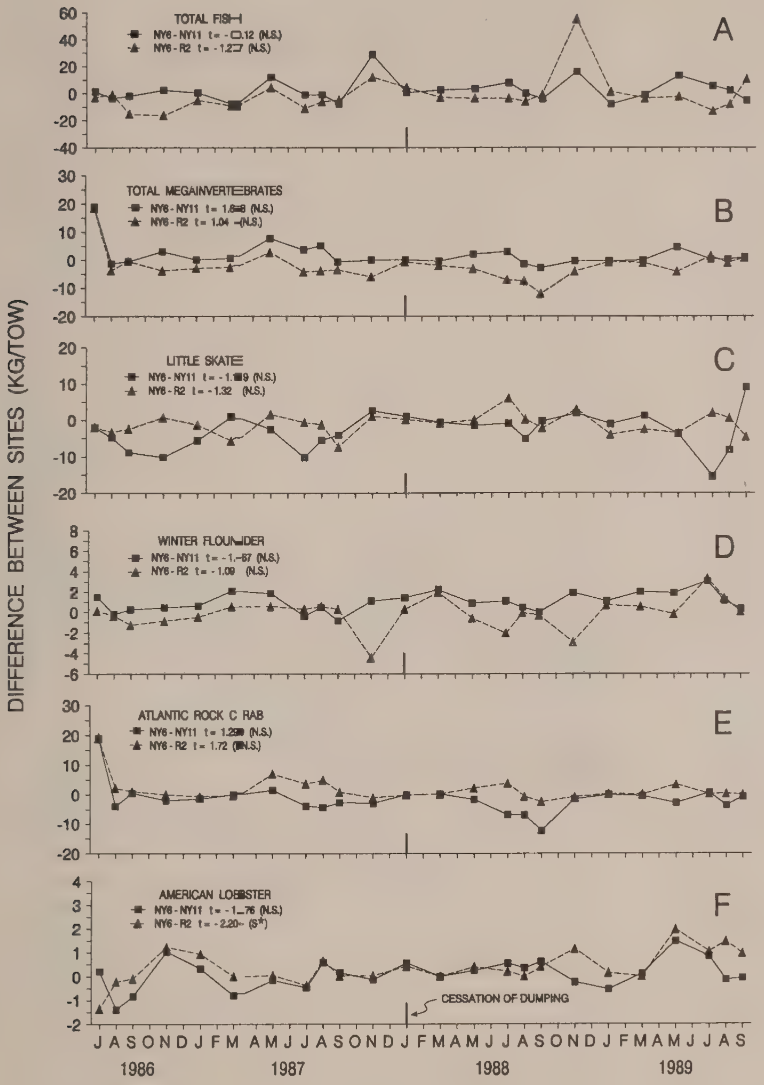

Figure 8

Replication in time of the difference in mean catch (kg/tow) between replicate stations sampled during a trawl survey in and around the 12-mile sewage dumpsite, July 1986–September 1989: A) total fish, B) total megainvertebrates, C) little skate, D) winter flounder, E) Atlantic rock crab, and F) American lobster. Triangles=NY6–R2, squares=NY6–NY11, N.S.=not statistically significant, and S*=statistically significant. See Table 3 for scientific names.

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Audience Questions

Question: I noticed that most of your tests are insignificant for individual species based on differences between the station you are looking at and some sort of reference station.

S. Wilk: Right.

Question: I wondered if you just looked at the absolute values themselves?

S. Wilk: Yes, we did. We looked at the values pre- and post[cessation], and they were not significant either. But in sticking with the statistical design, the test that we are using is one in which we subtract the two other stations from the impacted station. We will put together all of the tests for all of the species over time and area later. It is basically a computer problem, but we did do the tests that you asked about, and there were no significant differences.

Question: If I understood your presentation correctly, your significance test was an increase after cessation of dumping in the difference between the two stations; is that correct?

S. Wilk: Yes. Basically we are speculating that fishing [for lobster] was redirected from one area to the other. One area went up while the other area went down.

Question: The redirection was much more increased at R2 rather than at NY6?

S. Wilk: Right. What you are seeing is a redirection of fishing effort; i.e. the movement of lobster pots from one area to the other. We caught more when they were not fishing; probably because we did not have to avoid pots.

R. Pikanowski: I think you are a little wrong there. There was not a redirection of effort; I think it was just new effort at NY6. The question then becomes whether putting baits in pots in the water attracts lobsters, making them more accessible to our trawl, or if they are actually removing some of the population that we would have caught. So depending on which of those cases is existent, either the test was highly significant or it was not, and we have not determined that yet.

Question: Are you suggesting it went up because of the bait attracting the lobster?

R. Pikanowski: No. We are not suggesting that. That is a possibility and we are not sure. It could also have been a result of the cessation of dumping. This is one of those confounding factors that is a weakness of the particular statistical design. In this case we have a way, perhaps, to filter it out. I do not know what kind of study we would run to do that, but we welcome suggestions.

Question: So were you using two-tail tests on all of these that we just saw?

R. Pikanowski: For the fish we were, because we had no prior reason to assume they would go up or down. As a matter of fact, if you looked at the *t*-statistics, they were all negative, meaning that there were more fish afterwards than before, except for rock crab, which decreased.

Question: Did you look at the winter flounder fishery? Were there any changes in effort?

S. Wilk: In terms of commercial or recreational fishery?

Question: Commercial.

S. Wilk: We do not have a measure of that really, as you are probably aware. I think Beth Phelan will address some of the winter flounder issues in her talk. In terms of whether there was more or less effort prior to or during, either commercially or recreationally, we do not have those numbers.

Question: That may be something to look at. You may not be talking to the right people. One other thing, relative to some of the tests. The very first month or two months in one of the graphs showed a significant decrease. It would be interesting to analyze the catches statistically and leave off that first month to see what would happen.

R. Pikanowski: What you are referring to is the information on rock crab and total megainvertebrates, which actually is a reflection of rock crabs. The first month we sampled, there was an incredible bloom of *Asabellides* (spaghetti mud), and we hypothesized that the crabs were there eating them. We caught 20 kg per tow, and our next highest catch during the rest of the survey was 10. You raise a good question. If *Asabellides* were blooming due to the presence of sludge, then that is a sludge effect, and we leave that point in the analysis. On the other hand, if *Asabellides* blooms are a random event and not related to sludge, then we can treat that as an outlier. In fact I ran the test both ways, but because of the high variability we still got a nonsignificant result.

The truth of the matter is that rock crab biomass decreased by one half regardless of whether you kept that point in or not. That gets back to whether such a change is ecologically or statistically significant, which rests on two different concepts. I would assume a decrease in biomass of half is ecologically pretty significant. We just could not prove it was statistically significant.

Question: A question I have goes back to some reviews of the literature that existed at the turn of the century and in the 1920's; when Jacot (1920 [On the marine mollusca of Staten Island, New York. Nautilus 23:111–115]) and others looked at mollusks and did a wide range of collecting around Staten Island and Raritan Bay. He had reported a 50% diminution in the numbers of mollusk species back in the '20's and attributed it to the increasing use of gasoline and diesel engines.

Bert Walford, who was your mentor, and mine also, talked about the numbers of species of fish that he thought should have occurred in the New York Bight and did not. I raise the question: what do you find in these trawls today? Are they a sort of junk fish that tend to live or be able to live in degraded habitats? The skates and winter flounder are fish that, relative to other fish, did not show the same incidence of finrot disease back when that was an issue. Can one hypothesize, perhaps, that the disappearance of things like sheepshead and the intermittency of sea trout or weakfish which used to be fairly abundant in our area at times is connected to the fact that the New York Bight has been in a state of degradation for over a half a century, maybe much longer?

The people who ran the New York Aquarium in 1912 had to move it from Battery Park out to Coney Island, simply because the water in the Hudson River in 1912 would not support fish in the aquarium. So one has to question what you are looking at now; is it real? One might expect some changes as water quality improves over the years. I do not know whether you want to try and crystal ball this one, but it is something that is in my mind when I look at these data.

S. Wilk: In looking at the groundfish catches for the last 25 years, I think we are seeing similar patterns of the flip-flop of the more "valuable" species, whether they be flounders, cod, haddock, or herring. Even down here in the Mid-Atlantic, where the primary species in the past were scup, winter flounder, silver hake, and summer flounder, we are seeing a flip-flop throughout a range of studies, with the elasmobranchs making up approximately 70% of the catch. In this area, they make up about 55 or 60%. I think it is indicative of what is happening along the coast. You would have to really look deep in your crystal ball to come up with an answer as to whether or not, if things keep on cleaning up, sheepshead will return to Sheepshead Bay. I think it is going to take a long time to answer that question, and I do not think anybody wants to support surveys on a monthly basis for the next five years with subsequent monitoring. But it certainly might show such changes. There are so many other variables that are affecting the distribution and abundance of these migratory animals. The fish are being exploited at different places at different times, and their recruitment processes are different from year to year. I think it is a very large puzzle that we may never unravel.

Question: Some races or subspecies (of fish) on the West Coast are thought to be affected by habitat degradation and are being declared endangered species. Do you think the Endangered Species Act might be a tool for looking at the crystal ball a little bit harder?

S. Wilk: I once spent a month in Washington reviewing legislation for [listing] the striped bass as an endangered species. While that was 15 or so years ago, it still was not declared an endangered or threatened species and the species has rebounded. By what mechanism? It depends on whether you are a manager or a scientist. Whether the Act would work or not I do not know. That will depend on how far down things are.

Patterns of Winter Flounder, *Pleuronectes americanus*, Abundance and Distribution in Relation to the 12-Mile Sewage Sludge Dumpsite in the New York Bight

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ABSTRACT

Winter flounder were present throughout the year in the 250 km^2 area around the 12-mile dumpsite. Abundance varied among stations and years with reduced catch in summer (August). Low summer catch was attributed to scattering and random dispersion. Winter flounder tagged and released in the dumpsite area exhibited movement to the surrounding embayments of New York and New Jersey and to points farther northeast. It is believed that the dumpsite winter flounder represent a mixture of inshore populations that are in transit, either randomly or in response to spawning periodicity.

Introduction

The New York Bight—bounded by the coasts of Long Island, New York, on the north, New Jersey on the west, and extending seaward to the edge of the continental shelf—has been a disposal site for a variety of waste materials for many years. Beginning in 1924, sewage sludge from the metropolitan New York-New Jersey area was dumped at a location 22.2 km off the New Jersey coast (hereafter referred to as the 12-mile dumpsite) in the inner Bight, with volumes increasing to a high of 8.3×10^6 wet tons in 1983. With cessation of dumping expected in December of 1987, a multidisciplinary study was initiated in 1986 to examine the response of habitat and biota to the cessation of dumping at the 12-mile dumpsite (EPD [Environmental Processes Division], 1988). Included in the study were several components concerning the potential effects on selected marine resource species. One of these species, winter flounder, *Pleuronectes americanus*, is economically important to both commercial and recreational fisheries in the Mid-Atlantic and was selected for special study as a potential indicator of sewage effects. Winter flounder, a demersal species, are in close contact with the sediment and forage associated with the dumpsite and could be subject to contaminant accumulation and its effects, e.g. disease. Since winter flounder are also migratory, determination of the extent to which they were utilizing areas influenced by sewage sludge during their movements was a principal focus of the study.

Although the movements of winter flounder along northeast coasts have been studied with considerable interest for many years (Lobell, 1939; Perlmutter, 1947; Saila, 1961; Howe and Coates, 1975; Van Guelpen and Davis, 1979), little was known of migrations in the inner New York Bight. Based on known patterns and movements in Long Island coastal waters, a working hypothesis was proposed that a significant portion of the winter flounder which utilize the 12-mile dumpsite were from Raritan Bay (EPD, 1988). To determine the patterns of distribution, a mark-recapture study was included in the overall investigation of the effects of cessation of dumping. Emphasis was placed on relating the abundance and distribution patterns found at the 12-mile dumpsite to the general movements and migrations of winter flounder in the area of the New York Bight.

Methods

The 12-mile dumpsite study consisted of two complementary sampling series: 1) a replicate survey; and 2) a broadscale survey, each conducted in alternate months except for August, when both were conducted to monitor rapid seasonal changes (Fig. 1A; EPD, 1988; Pikanowski, 1995). In the replicate survey, repeated measurements (n=8) were made for a suite of variables at three stations (R2, NY6, NY11), which were similar bathymetrically, but which represented different levels of sewage sludge accumulation and effects. The broadscale survey consisted of single, nonreplicated measurements at 22 stations, covering an area of 250 km^2 across the inner Bight, and included all major habitat types (EPD, 1988).

From August 1986 through August 1989 winter flounder were collected every other month at the 22 broadscale stations (Fig. 1A). In addition, supplemental cruises were made periodically during even numbered months throughout the inner Bight to increase the number of winter flounder collected for tagging. In a concurrent study, fish were sampled at 14 inshore sites in Sandy Hook, Lower and Raritan Bays, and the lower Shrewsbury River; these sites will be collectively referred to as Raritan Bay (Phelan, 1992).

Winter flounder were caught using an otter trawl (11.0-m footrope and 9.8-m headrope with a 51-mm stretched mesh 18-thread knotted nylon cod end) towed for 15 min at each location at approximately 5.6 km/hr (Wilk et al., 1995). Water temperature was measured at each location 1.0 m above the bottom with a Beckman RS–5 temperature-salinity meter or with a thermometer reading taken from a bottom-tripped Niskin bottle. Following capture, winter flounder were held in a container with flow-through seawater until processed. The total lengths (TL) of all winter flounder were measured to the nearest millimeter. Although all sizes of winter flounder were included in the length frequency distributions, only fish ≥18 cm were selected for tagging. Danila (1978) showed that over half the male winter flounder and up to one third of the females sampled in a southern New Jersey estuary were mature during their first winter (18.0–24.0 cm). Therefore, tagging of fish ≥18 cm increased the chance of recapturing mature individuals, which are believed to be the principal migrants (McCracken, 1963). Winter flounder were tagged with yellow laminated plastic Peterson discs (1.27 cm in diameter) imprinted with an identification number, return address, and a request for catch information. The tag was attached with a nickel pin through the dorsal musculature and was held by a crimp in the pin against a blank disc on the opposite side. Fish were released in the general vicinity of the capture site. The tagging program was publicized via a press release and in local recreational fishing publications. A monetary reward for returned tags was not offered. The length frequency information collected from all winter flounder caught during the broadscale survey was combined and plotted by month combining data across years.

Results

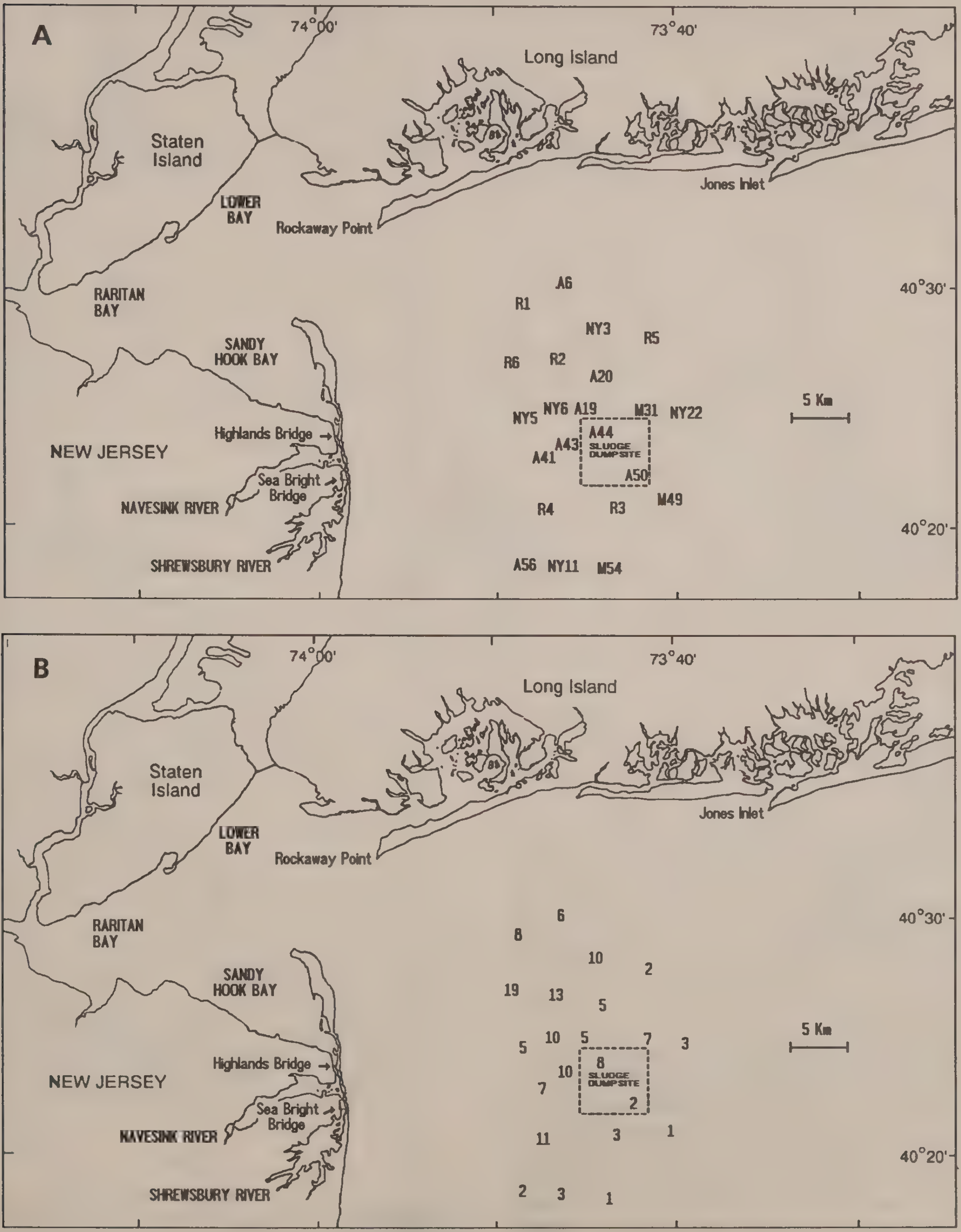
From August 1986 to August 1989, 2,392 adult (≥18 cm) and 240 juvenile winter flounder were caught during the broadscale survey of the 22 offshore dumpsite stations. In addition, 842 adult winter flounder were caught and tagged on supplementary dumpsite cruises. Winter flounder were caught in all broadscale sampling months (Fig. 2). The number of winter flounder caught per month did not appear to be correlated with temperature although there was a trend towards reduced catch in summer (August). During other months, catches were larger but fluctuated widely.

The largest dumpsite survey catches of winter flounder came from stations R2 and R6 (Table 1; Fig. 1). While the abundance of winter flounder per station varied, certain stations east and south of the dumpsite center, i.e. R5, A50, M49, R3, A56, NY11, NY22, and M54, yielded consistently low average catches (Fig. 1B). Conversely, stations north and west of the dumpsite center (NY3, R2, R6, NY6, and A43) had higher average catches. These trends in abundance were not apparently related to the water depth of the sampling area (Table 1).

The length frequencies of winter flounder caught in the dumpsite area ranged from 11.0 to 43.0 cm (Fig. 3). Fish collected for tagging (639) in non-broadscale months were excluded from the figure. Juveniles, which first appeared in December, formed a second distinct size group. Both juvenile (<18 cm) and adult (≥18 cm) cohorts appeared in February, April, and June. With growth, the juveniles were assimilated into the adult population resulting in a smaller modal adult size from August through October. During February and April when the juveniles formed a separate cohort, the adult modal size was 25.0 cm, shifting to 26.0 cm in June.

To date, of the 3,234 fish tagged at the dumpsite, 44 winter flounder have been recaptured for a return rate of 1.4% (Table 2). The majority of the returns (81.8%) were made by recreational fishermen, followed by research vessels (9.1%), and commercial vessels (9.1%). The number of days at large ranged from 1 to >800. The majority of fish (82%) were at large >100 days before recapture. There was no significant correlation between the size of the fish and the number of days at large (r=0.01, n=44).

The largest number (31) of winter flounder returns were from locations west of South Oyster Bay, Long Island (Table 2; Fig. 4). Fish were recaptured from sites 45 km northeast of the center of the dumpsite at South Oyster Bay, Long Island, to sites 35 km southwest of the dumpsite at Shark River, N.J., and at sites in waters northwest up to and including Raritan Bay (Fig. 4). Fish were recovered primarily during March, April, and May from waters in Raritan Bay (Great Kills, Staten Island;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74°00'
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Long Island
Staten Island
LOWER BAY
Rockaway Point
Jones Inlet
RARITAN BAY
SANDY HOOK BAY
Highlands Bridge
NEW JERSEY
Sea Bright Bridge
NAVESINK RIVER
SHREWSBURY RIVER
40°30'
40°20'
5 Km
A6
R1
NY3
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R6
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NY5
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SLUDGE DUMPSITE
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SANDY HOOK BAY
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Figure 1

A) Winter flounder, *Pleuronectes americanus*, tagging sites in the inner New York Bight, August 1986–August 1989. B) Average number of winter flounder per tow from August 1986–August 1989 at each tagging site shown in A.

Table 1

Total number of winter flounder (*Pleuronectes americanus*) caught during the broadscale survey of the 12-mile dumpsite per station per year and average number of winter flounder per tow from August 1986-August 1989 (see Fig. 1).

		Year				Flounder collected (n)	
		1986	1987	1988	1989		
		months sampled (n)					
Station	Depth (meters)	(3)	(6)	(6)	(4)	Total	$\bar{x}$/tow
A6	23.9	7	44	29	32	112	5.9
R1	25.5	4	66	42	37	149	7.8
NY3	26.2	18	83	69	22	192	10.1
R5	21.6	5	5	14	7	31	1.6
*R2	28.8	18	91	76	38	223	13.1
R6	29.9	38	176	98	56	368	19.4
A20	28.2	4	19	35	33	91	4.8
*NY6	30.9	17	43	32	82	174	10.2
A19	26.1	14	24	37	27	102	5.4
M31	24.1	16	30	41	39	126	6.6
NY5	34.6	17	47	12	18	94	4.9
NY22	23.9	7	16	18	18	59	3.1
A44	23.8	27	33	46	47	153	8.1
A43	26.8	43	61	59	28	191	10.1
A41	37.8	12	48	24	44	128	6.7
A50	24.4	1	14	13	15	43	2.3
M49	24.7	3	1	5	14	23	1.2
R3	25.9	9	15	11	21	56	2.9
R4	32.5	34	78	57	40	209	11.0
A56	55.8	15	9	11	1	36	1.9
*NY11	30.5	5	4	25	17	51	3.0
M54	25.9	3	2	6	3	14	0.7

* R2, NY6 and NY11 were only sampled five months for the broadscale surveys during both 1987 and 1988. Means per tow are therefore based on 17 tows.

Lower Bay; lower Shrewsbury River, N.J.) and the southern bays of western Long Island (Jamaica Bay; Sheepshead Bay; Baldwin Bay) (Table 2). Four flounder were recaptured in the nearshore coastal waters of western Long Island (Rockaway Beach and Jones Beach). Finally, one winter flounder was recovered southwest of the dumpsite at Shark River, N.J. (Fig. 4A).

Only four winter flounder recaptures came from within the broadscale area around the 12-mile dumpsite (Fig 4a, Table 2). One of these fish was recaptured in the same location (NY6) almost two years after it was originally tagged. Two other fish were at large for about a year and were caught within the broadscale area but not at the tagging location. One dumpsite fish was tagged at R1 on one day and captured nearby at Ambrose Light the next day (3.5 km).

Thirteen tagged fish exhibited movement more than 45 km to the northeast of the dumpsite (Fig. 4B) with returns recorded from the waters of Fire Island, N.Y.; Long Island Sound; Montauk Point, Long Island; Block

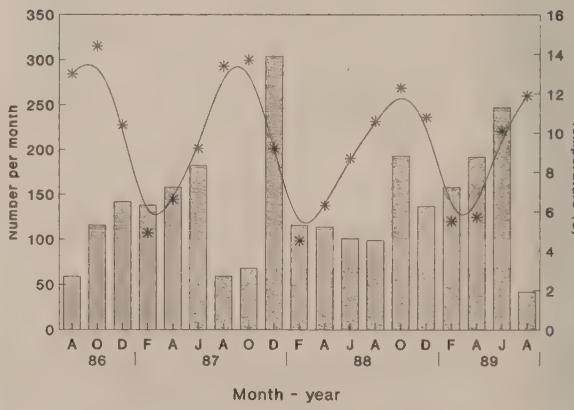

Figure 2

Total number of winter flounder, *Pleuronectes americanus*, caught per month (bars) and average monthly bottom temperature (stars) during the sampling of the inner New York Bight, August 1986–August 1989. Trend line represents the best fit of the available temperature data.

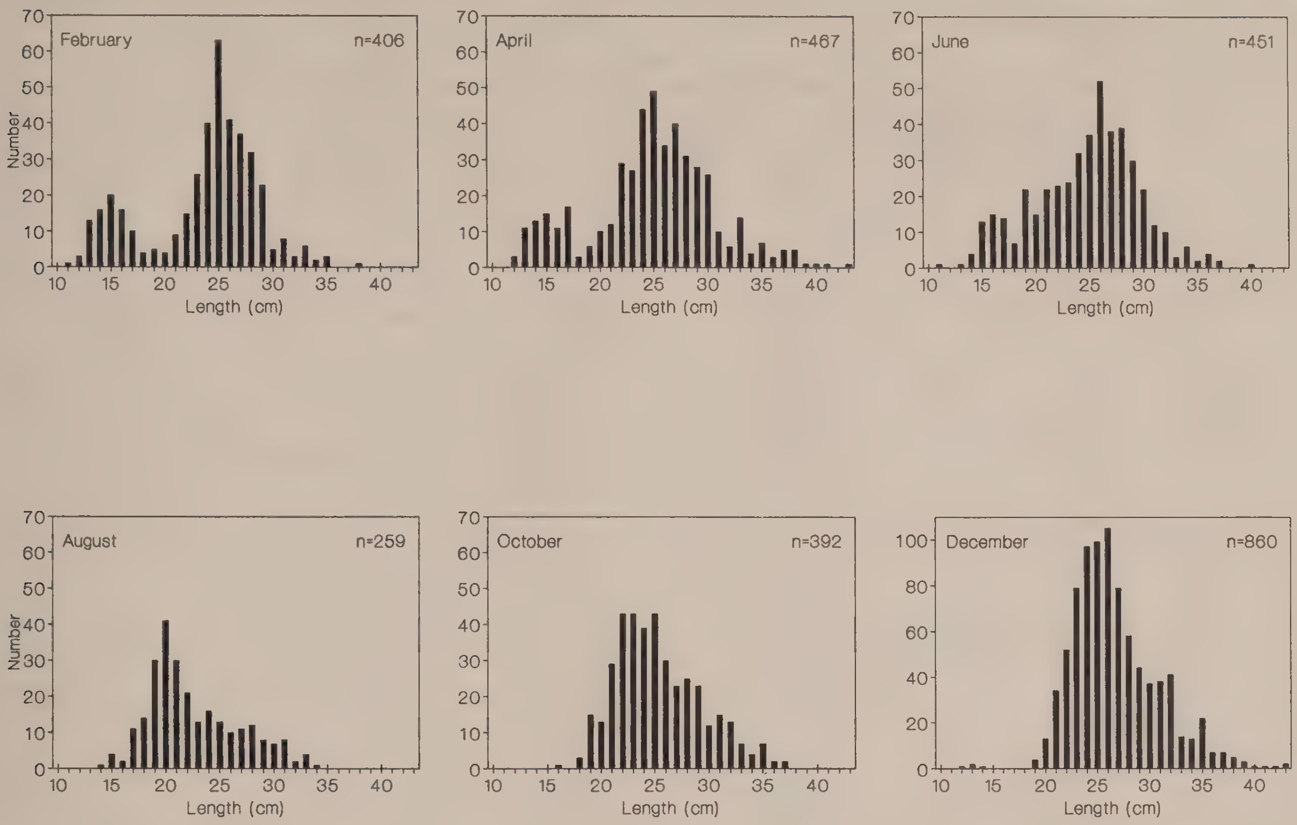

Figure 3

Length frequency distributions of winter flounder, *Pleuronectes americanus*, collected from the inner New York Bight broadscale survey, July 1986–August 1989, plotted by month combining data across years.

Island, south of Rhode Island, and Nantucket Shoals, south of Massachusetts (Table 2, Fig 4b). There was no significant correlation between the size of fish tagged at the dumpsite and distance travelled (estimated straight line) to the recapture point (r=0.01, n=44).

In the concurrent study, 4,101 flounder were tagged at inshore stations; the majority of fish were recaptured from a variety of locations in Raritan Bay and the south shore of Long Island (Phelan, 1992). These patterns of movement substantiate that winter flounder exhibit seasonal spawning migrations and the ability to return to a natal site to spawn. The tag returns also revealed a degree of "wandering" with fish recovered from Shark River, N.J., and Nantucket Shoals, Mass.

Discussion

Winter flounder were abundant year round in the broadscale area around the 12-mile dumpsite (Wilk et al., 1995). Generally, winter flounder catches were greater at the northwestern broadscale stations. Oviatt and Nixon (1973) found by multiple regression analysis that winter flounder abundance and distribution were correlated with temperature, depth and sediment organic content in Narragansett Bay, R.I. However, in the present study, the between-station variability in catch did not exhibit any discernable relation with average bottom temperature or depth. Sediment organic content, as measured by total organic carbon, was highest at two northern stations (R2 and NY6) prior to cessation but decreased significantly thereafter (Packer et al., 1995). It is possible that winter flounder were less abundant at the southern stations not because of suboptimal habitat but because the northern stations lie within the migratory path used by winter flounder as they move through the New York Bight.

Adult winter flounder collected at the broadscale stations varied in abundance throughout the year, consistently showing a temporal pattern of reduced numbers in summer (August). This agrees, in part with Howe and Coates (1975), who found that southeast of Cape Cod, winter flounder released in the spring and summer spread farther offshore, returning to shoal areas during fall and winter. Saila (1961) also reported that winter flounder disperse during the summer months

Table 2

Winter flounder, *Pleuronectes americanus*, tag returns from fish captured and released in the area of the 12-mile dumpsite, July 1986 - August 1989.

Length (cm)	Tagging Location	Tagging Date	Recapture Date	Recapture Location	Days at Large
Dumpsite to Sandy Hook-Raritan Bays and Rivers					
21.0	R2	18 Jul 86	13 Apr 87	Lower Bay	275
26.3	R2	27 Jan 87	? Apr 87	Lower Bay	108
18.2	NY6	18 Feb 87	28 Apr 89	Lower Shrewsbury River	800
36.6	R6	10 Jun 87	1 May 88	Lower Bay	326
29.9	R6	9 Dec 87	29 Apr 89	Lower Bay	142
32.0	R6	21 Dec 87	18 Mar 88	Lower Shrewsbury River	90
31.0	R4	27 Oct 88	9 Apr 89	Lower Bay	164
30.2	A41	9 Dec 88	10 Apr 89	Lower Shrewsbury River	122
28.3	R2	22 Dec 88	15 Apr 89	Lower Bay	114
Dumpsite To Western Long Island, New York					
27.0	A43	17 Oct 86	27 Mar 87	Jones Inlet, LI	161
23.0	R6	14 Apr 87	27 Mar 88	Seaford, LI	348
24.0	NY5	15 Apr 87	28 May 87	Ocean off Rockaway, LI	43
26.5	NY3	9 Jun 87	29 Mar 88	Jamaica Bay, LI	294
23.8	R6	19 Oct 87	25 Oct 88	Reynolds Channel, LI	342
24.0	NY3	9 Dec 87	12 Dec 87	Jones Inlet, LI	3
26.7	R6	9 Dec 87	20 Apr 88	Jamaica Bay, LI	133
24.6	NY3	21 Dec 87	9 Apr 88	Sheepshead Bay, LI	110
25.5	NY3	21 Dec 87	14 Apr 89	Massapequa, LI	115
31.8	R6	21 Dec 87	1 May 90	Sheepshead Bay, LI	862
29.7	R2	22 Dec 87	9 May 88	South Oyster Bay, LI	139
22.6	R6	22 Feb 88	24 Nov 88	Ocean off Jones Inlet, LI	276
27.3	R6	26 Apr 88	29 Mar 89	Baldwin Bay, LI	337
28.8	A43	8 Jun 88	9 May 90	Jones Beach, LI	700
27.5	R1	11 Oct 88	25 Mar 89	Jamaica Bay, LI	165
32.9	R6	21 Dec 88	1 Jun 89	Ocean off Rockaway Beach, LI	162
27.5	M31	7 Feb 89	29 May 89	Ocean off Fire Island, NY	111
24.7	M31	7 Feb 89	12 Jun 89	Ocean off Fire Island, NY	111
35.0	R4	13 Feb 89	1 May 89	Ocean off Coney Island, NY	77
Dumpsite to Eastern Long Island, New York and farther north					
26.8	NY22	25 Feb 87	8 May 87	Montauk, LI	72
25.9	M31	25 Feb 87	8 Aug 87	Ocean off Shinnecock, LI	164
24.9	NY6	29 Jan 88	19 May 88	Ocean off Shinnecock, LI	111
24.7	NY6	10 Feb 89	13 May 89	Shinnecock Bay, LI	92
27.5	A19	18 Feb 87	23 Oct 87	Nantucket Shoals	247
27.0	NY6	15 Apr 87	13 Jul 87	South of Martha's Vineyard, MA	104
28.8	R2	21 Dec 87	9 May 88	Long Island Sound	140
31.5	NY11	10 Jun 88	20 Apr 89	Shark River, NJ	314
22.5	NY3	6 Feb 89	11 May 89	SW of Block Island, RI	94
27.7	R6	10 Feb 89	3 Mar 90	East of Block Island, RI	386
29.2	NY6	10 Feb 89	22 Jun 90	South of Block Island, RI	497
35.0	R4	13 Feb 90	8 Jan 90	South Martha's Vineyard	329
Dumpsite to Dumpsite					
34.6	NY6	15 Apr 87	10 Feb 89	NY6	667
27.5	R1	7 Dec 87	8 Dec 87	Ambrose Light	1
25.4	R1	8 Feb 88	22 May 89	A56	469
25.9	R6	11 Feb 88	30 Dec 88	A6/NY3	323

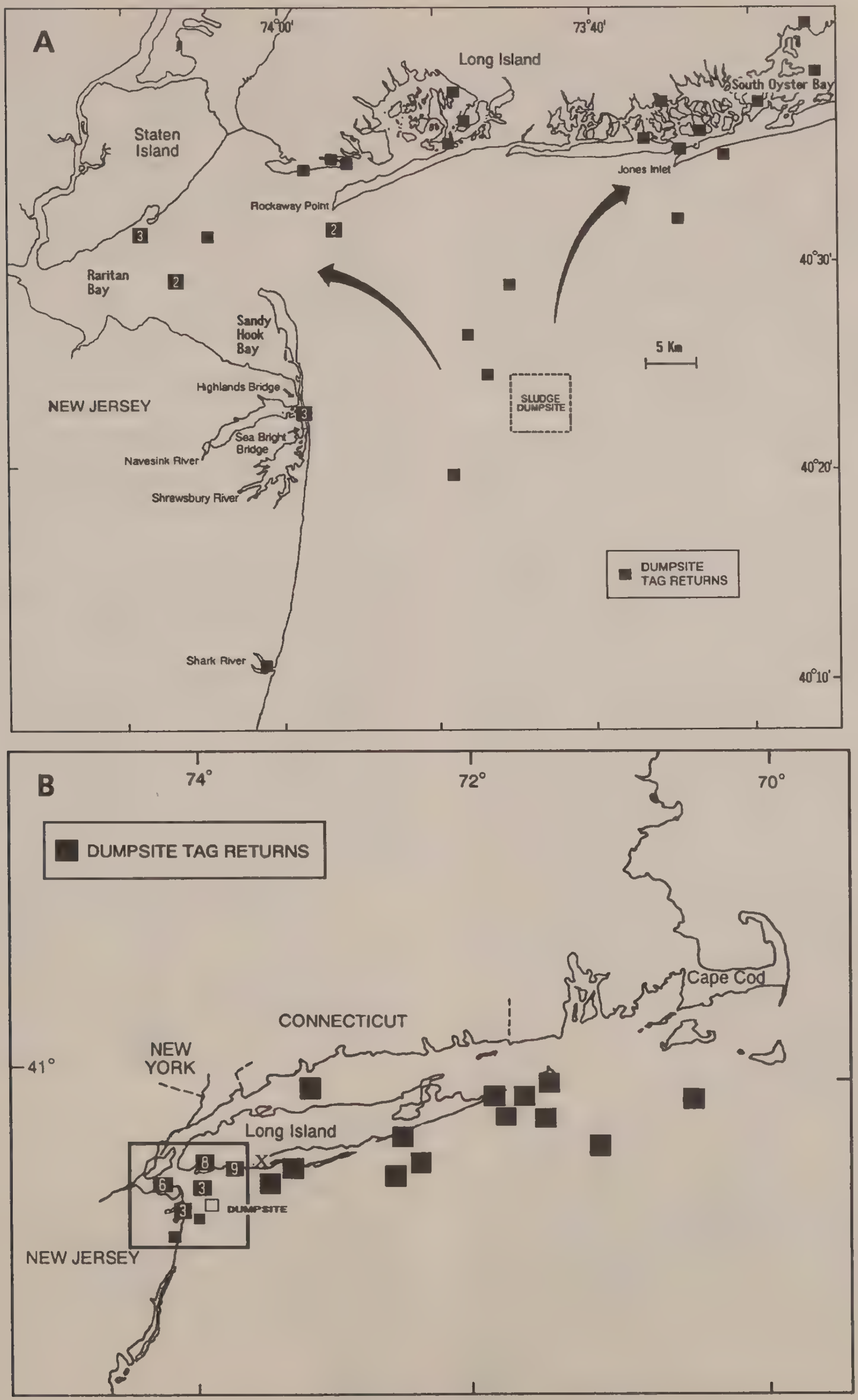
A
74°00'
73°40'
Long Island
South Oyster Bay
Staten Island
Jones Inlet
Rockaway Point
3
2
40°30'
Raritan Bay
2
Sandy Hook Bay
5 Km
Highlands Bridge
SLUDGE DUMPSITE
NEW JERSEY
3
Sea Bright Bridge
Navesink River
40°20'
Shrewsbury River
DUMPSITE TAG RETURNS
Shark River
40°10'
B
74°
72°
70°
DUMPSITE TAG RETURNS
Cape Cod
CONNECTICUT
NEW YORK
41°
Long Island
8
9
X
6
3
3
DUMPSITE
NEW JERSEY

Figure 4

A) Winter flounder, *Pleuronectes americanus*, tag returns west of South Oyster Bay, Long Island, in the inner New York Bight, August 1986–August 1989. Each symbol represents one return unless indicated by an inscribed number. B) Winter flounder tag returns from the New York Bight, highlighting tag returns from locations east of South Oyster Bay, Long Island (indicated by an X), August 1986–August 1989. Each symbol represents one return unless indicated by an inscribed number.

Disease Prevalence of Inner New York Bight Winter Flounder, *Pleuronectes americanus*, Collected During the 12-Mile Dumpsite Recovery Study, 1986–1989

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ABSTRACT

The prevalence and distribution of diseases and abnormalities possibly associated with municipal waste pollution in the New York Bight are described from observations of winter flounder, *Pleuronectes americanus,* samples collected at three stations representing different levels and effects of sludge influence from 1986 to 1989. There were reductions in incidence of fin rot and internal lesions, conditions possibly associated with facultative pathogens, following the cessation of sewage sludge dumping. No significant differences were evident in the prevalence of the stress-provoked infections of lymphocystis (viral) and *Glugea stephani* (microsporidial). No differences related to the cessation of sludge dumping were noted in prevalence of the somatic abnormalities, bentfin, and ambicoloration.

Introduction

Sludge from approximately 200 sewage treatment plants was dumped at a site in the New York Bight 12 nmi from Sandy Hook (Fig. 1) since 1924. By the early 1980's the volumes of sludge dumped were larger than at any other sludge dumpsite in the world (Norton and Champ, 1989). This disposal activity was phased out at the end of 1987. The cessation of ocean dumping at this nearshore site offered an opportunity to document the response of an area to the removal of a major waste loading. The Environmental Processes Division (EPD) of NOAA's Northeast Fisheries Science Center (NEFSC) conducted a multidisciplinary study of the nearshore sludge dumping area from summer 1986 to fall 1989 (EPD, 1988; Studholme, 1988).

The objective of this part of the study was to determine if detectable changes in the prevalence of gross pathology in selected species, eventually limited to winter flounder, *Pleuronectes americanus,* occurred during the phaseout and postdumping periods. Several authors concluded that prevalence of disease serves as a sentinel of water quality (Sindermann, 1979, 1985; Murchelano, 1988). They have reported that appearance or increase in the incidence of a number of diseases, including fin erosion, ulcers, possibly lymphocystis in fishes, chitinoclasia in crustaceans, and certain neoplasms in bivalve mollusks, has been associated with degraded coastal water quality. In this study, the premise to be tested was whether reduction of a pollutant stress, i.e. sewage sludge, results in a measurable decrease in the prevalence of diseases attributed to facultative patho-

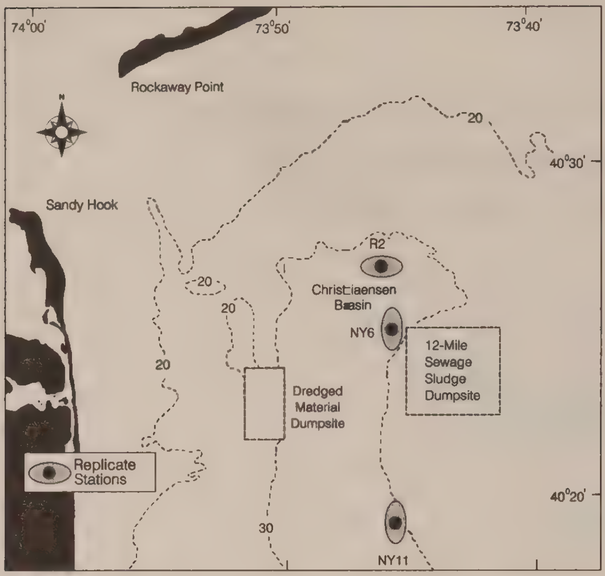

Figure 1
Location of replicate stations sampled for disease prevalence in winter flounder, *Pleuronectes americanus*, during the 12-mile dumpsite study, 1986–89. Isobaths are given in meters.

gens, stress-provoked latent infections, or abnormalities induced environmentally.

Methods

The 12-mile dumpsite study (12-MDS) was designed around two complementary sampling series, one replicated at three reference stations (NY6, R2, and NY11), the other a broadscale survey of single stations sampled over approximately 350 km^2 (Fig. 1) (EPD, 1988; Pikanowski, 1995). Each was conducted in alternate months except for August, when both were done. During the replicate survey, eight trawl tows were made at each of three stations, using an otter trawl with an 11-m footrope and cod end of 2-inch stretch mesh (Wilk et al., 1995). Trawl collections of the bottomfish community were made routinely to measure relative abundance, biomass, and length frequency. The three replicate survey stations were bathymetrically similar, but with different histories of sewage accumulation. Station NY6, the most heavily degraded, was approximately 1.6 km downslope from the northwest corner of the dumpsite, the section receiving the greatest amount of sludge. Station R2, 3.4 km north of the dumpsite, was in a pollutant-enriched area on the north edge of the Christiaensen Basin. Station NY11, 10 km SSW of the dumpsite epicenter, was on the eastern shoulder of the Hudson Shelf Valley and considered the least polluted of the three.

Winter flounder, a ubiquitous demersal species, was selected for both food habit and gross disease prevalence studies. Observations of disease prevalence began September 1986 and continued until the end of sampling in September 1989. All winter flounder collected during the replicate surveys were sacrificed and shipboard dissection procedures involving stomach removal (Steimle, 1995) provided opportunities for internal and external detection of gross pathology. Coelomic organs were inspected for the presence of lesions and xenomas of *Glugea stephani*, an intestinal parasite. External observations were made for fin rot, bentfin, lymphocystis, and ambicoloration.

Disease observations were partitioned into three time periods. The first, 1986–87, includes observations from the precessation period, when dumping was still occurring at a reduced "phase-out" level. The second and third periods, 1988 and 1989 respectively, include observations made after dumping stopped. For each of the three periods, disease prevalence (ratio of diseased to total in catch) among the three replicate stations was tested statistically by a *G*-test of significance (Sokal and Rohlf, 1981), which tests the goodness-of-fit between observed and expected frequencies (ratios). Data were entered into a BIOM program, and columns and rows of the contingency tables were tested for independence. Observed values were tested against hypothetical values expected if there were no differences among periods or among stations. *G* values were compared with a critical value of chi-square at the 95% level of significance; if *G* values were below the critical value the null hypothesis was accepted, i.e. there was no statistical evidence to suggest sampling sites or periods were significantly different. To develop an index of change from the dumping period to subsequent years, observations from all stations were grouped by period and expressed as means. Chi-square tests also were made on the combined station totals of dumping and postdumping periods.

Results

From the 3,919 winter flounder examined (Table 1), we observed examples of disease in three general categories; 1) those likely attributable to facultative pathogens, such as fin rot and internal lesions (visible foci of inflamed tissue); 2) those resulting from stress-provoked latent infections, such as lymphocystis or *Glugea stephani*; and 3) non-infectious types such as the somatic abnormalities of ambicoloration and bentfin.

Fin rot

Fin rot is a disease characterized by destruction of epidermal fin tissue (Post, 1987). McCain et al. (1982)

Table 1

Numbers (n) of winter flounder, *Pleuronectes americanus*, examined during the 12-Mile Dumpsite Study, 1986–1989. Data are grouped by replicate station and by phase-out (1986–87) and postdumping (1988–1989) periods.

	Station			
Period	NY6	R2	NY11	n
1986–87	441	586	217	1,244
1988	577	816	233	1,626
1989	585	286	178	1,049

described the condition idiopathic in Puget Sound waters because they found no pathogenic bacteria or viral inclusions associated with areas of fin erosion on either English sole, *Parophyrys vetulus*, or starry flounder, *Platichthys stellatus*. Mahoney et al. (1973) however, found a gram negative bacteria assocation with fin rot on a variety of species, including winter flounder, taken from inner waters of the New York Bight. The etiology of fin rot in wild marine fish remains largely unknown. Although all fins may be involved, the disease appeared most frequently on the dorsal and anal fins. It has been documented in the New York Bight since 1973 and monitored almost continuously since (Mahoney et al., 1973; Murchelano, 1975; Ziskowski and Murchelano, 1975; Murchelano and Ziskowski, 1976; 1979, 1982; Sindermann, 1979; O'Connor et al., 1987). Mean annual fin rot prevalence among winter flounder from the apex was summarized by O'Connor et al. (1987) (Table 2). The data suggest fin rot incidence in the Bight has diminished generally from 1973 to 1983.

Fin rot prevalence observed in the present study and results of *G*-tests are presented in Table 3. The presence of fin rot at NY11 during dumping followed by none observed in subsequent years resulted in a statistically significant *G*-value (8.55) and rejection of the null hypothesis for that location. A progressive temporal reduction in fin rot prevalence was also observed at station R2. The computed *G*-value (5.13) approached, but did not exceed, the 95% confidence level of significance. Fin rot prevalence at NY6 also decreased following cessation of dumping, but the change was not statistically significant. There was no significant difference among the three stations by year. The time trend of grouped station-weighted prevalence means (1.04:0.36:0.10) suggests a general reduction of fin rot to levels less than those reported in 1983. The prevalence estimates for the 1986–87 dumping period were significantly higher ($P<0.001$) than those for the postdumping period.

Table 2

Summary of fin rot prevalence in winter flounder, *Pleuronectes americanus*, from the New York Bight (Source: O'Connor, et al. 1987).

Year	Percent fin rot	Sample size
1973	13.40	1,943
1974	6.14	570
1975	1.59	1,637
1976	0.69	667
1977	3.19	1,159
1978	2.03	2,561
1979	—	—
1980	0.00	102
1981	1.59	314
1982	1.68	357
1983	0.41	241

Table 3

Fin rot prevalence (%) and results of G-tests for winter flounder, *Pleuronectes americanus*, examined during the 12-Mile Dumpsite Study, 1986–1989. Critical values of Chi-square are 5.99 at the 95% level of confidence and 9.21 at the 99% level. G-values exceeding the 95% level are denoted by *, those exceeding the 99% level by **.

	Station			
Period	NY6	R2	NY11	G-value
1986–87	0.91	0.85	1.84	1.40
1988	0.69	0.24	0.0	3.42
1989	0.17	0.0	0.0	1.17
G-value	3.14	5.13	8.55*	

Internal lesions

During the dissection process we noted occasional foci of lesions, angry reddened areas of inflammation, most often on either liver or ovary. We did not histologically characterize these lesions to determine their etiological basis. Prevalence estimates of these lesions and results of *G*-test (Table 4) indicate highly significant declines at all three stations after dumping ceased. The combined data for all stations indicate a decrease in lesion prevalence from 4.44% during the dumping period, to 0.86% in 1988 and 0.38% in 1989. The 1986–87 prevalence estimates were significantly different ($P<0.001$) from those of the postdumping period. In a description of cysts and ulcers in fish, Sindermann (1979) stated that a vast and almost unmanageable literature about induced lesions exists, but all data derive from laboratory exposure studies. It may be conjec-

Table 4

Internal lesion prevalence (%) and results of G-tests for winter flounder, *Pleuronectes americanus*, examined during the 12-Mile Dumpsite Study, 1986–1989. Critical values of Chi-square are 5.99 at the 95% level of confidence and 9.21 at the 99% level. G-values exceeding the 95% level are denoted by *, those exceeding the 99% level by **.

	Station			
Period	NY6	R2	NY11	G-value
1986–87	4.08	3.58	7.37	4.92
1988	1.04	0.86	0.43	0.83
1989	0.17	1.04	0.0	4.49
G-value	29.13**	25.76**	14.39**	

Table 5

Lymphocystis prevalence (%) and results of G-tests for winter flounder, *Pleuronectes americanus*, examined during the 12-Mile Dumpsite Study, 1986–1989. Critical values of Chi-square are 5.99 at the 95% level of confidence and 9.21 at the 99% level. G-values exceeding the 95% level are denoted by *, those exceeding the 99% level by **.

	Station			
Period	NY6	R2	NY11	G-value
1986–87	0.68	1.19	0.46	1.35
1988	0.17	0.61	0.43	1.71
1989	0.17	0.35	0.0	1.00
G-value	1.33	2.31	2.30	

tured that the coincident drop in contaminant levels after dumping stopped, particularly heavy metals (Zdanowicz et al., 1995), may be a contributing factor to results observed in the field.

Lymphocystis

Gross lesions of this common viral disease were easily recognized. Presence of the lymphocystis virus induces connective tissue cells to increase in mass and the resulting giant cells appear commonly as nodules beneath the epidermis, especially on fins (Russell, 1974). Lymphocystis is widely distributed (Post, 1987) and noted in winter flounders from the New York Bight. Prevalence of 2.35% in the inner New York Bight area was reported by Ziskowski et al. (1987), ranking second of eight areas from Cape May to the Canadian Border; highest prevalence was 4.46% in the offshore New York Bight (depth range of 27–200 m). Studies in other areas associated increased prevalence of lymphocystis with degraded environments (Murchelano, 1982). Prevalence estimates of lymphocystis and results of G-test (Table 5) indicate there were no significant differences in lymphocystis infections either among periods or among stations. Our values were all markedly lower than the mean prevalence of 2.35% reported by Ziskowski et al. (1987). The weighted mean prevalence of grouped stations trended downward from 0.88% during the precessation interval to 0.42% in 1988 and 0.20% in 1989. However, the 1986–87 dumping prevalence estimates were not significantly different from those of the postdumping period.

Glugea stephani

Infections of the microsporidean protozoan, *Glugea stephani*, are well documented in winter flounder. The tumorlike masses of developing spores are easily observed in the intestinal tracts. Infections from southern Massachusetts have been known and described since the turn of the century; a 50% infection rate was noted in winter flounder from the Woods Hole area in 1910 (Mavor, 1915). The range of this protozoan was extended south, to the New York Bight, by Takvorian and Cali (1981). Takvorian and Cali (1984) reported a prevalence of infection from 0.63% in October 1978 to 25.0% in August 1980 in specimens taken in estuarine waters of the New York–New Jersey Lower Bay Complex. A significant increase occurred during summer, with a peak in August. We have found no references that attempt to correlate degraded water environments with infection rate of *Glugea stephani*.

The only significant departures from expected prevalences of *Glugea stephani* were those observed among periods at station NY11 (Table 6). The high frequency observed at NY11 and, to a lesser degree, at NY6 in 1988, albeit significant (rejecting H_o) has no assignable cause. Water temperature has been noted as the single most significant factor in the occurrence of this disease (McVicar, 1975), but midsummer temperatures of bottom water in 1988 were 3–4°C less than the previous and following years. The weighted mean prevalences of *Glugea stephani* at all stations for successive time periods were 2.25:4.15:1.34. There was no significant difference (*P*=0.8) between the 1986–87 prevalence estimates and those of the postdumping period.

Ambicoloration

Aberrant pigmentation patterns of melanistic patches on the blind side of winter flounders were noted. Our

Table 6

Glugea stephani prevalence (%) and results of G-tests for winter flounder, *Pleuronectes americanus*, examined during the 12-Mile Dumpsite Study, 1986–1989. Critical values of Chi-square are 5.99 at the 95% level of confidence and 9.21 at the 99% level. G-values exceeding the 95% level are denoted by *, those exceeding the 99% level by **.

	Station			
Period	NY6	R2	NY11	G-value
1986–87	2.04	2.90	0.92	3.36
1988	2.77	1.77	4.29	5.01
1989	1.03	1.75	1.6	0.79
G-value	7.03*	4.98	2.44	

Table 7

Ambicoloration prevalence (%) and results of G-tests for winter flounder, *Pleuronectes americanus*, examined during the 12-Mile Dumpsite Study, 1986–1989. Critical values of Chi-square are 5.99 at the 95% level of confidence and 9.21 at the 99% level. G-values exceeding the 95% level are denoted by *, those exceeding the 99% level by **.

	Station			
Period	NY6	R2	NY11	G-value
1986–87	0.45	1.02	1.38	1.79
1988	1.39	0.49	1.29	1.53
1989	2.22	2.10	1.68	0.20
G-value	0.11	6.30*	5.24	

totals do not include the occasional but rarer examples of partial albinism (nonpigmented areas on the eyed side) (Dawson, 1971). Ziskowski et al. (1987) reported a mean prevalence of 0.39% for the New York Bight, second only to southern New England (0.89%). We found no significant differences among stations for any of the three time intervals (Table 7). There were differences indicated over time at NY6 and R2, along with a weak trend of increase with the highest prevalence at each station occurring in 1989. The weighted means of ambicolor prevalence of the grouped station data were 0.88%:0.92%:2.09%. The significant increase in prevalence from the precessation to postdumping period (P<0.001) is unexplainable, but is unlikely to be related to dumpsite influence.

Table 8

Bent finray prevalence (%) and results of G-tests for winter flounder, *Pleuronectes americanus*, examined during the 12-Mile Dumpsite Study, 1986–1989. Critical values of Chi-square are 5.99 at the 95% level of confidence and 9.21 at the 99% level. G-values exceeding the 95% level are denoted by *, those exceeding the 99% level by **.

	Station			
Period	NY6	R2	NY11	G-value
1986–87	1.59	2.56	1.84	1.25
1988	2.25	1.96	2.15	0.15
1989	2.05	1.74	0.56	2.27
G-value	1.99	0.60	0.73	

Bentfin

This condition, characterized by a bending of the midportion of a series of finrays, was noted as an obvious line of flexure, or folding, of a fin. The condition was described from winter flounder taken in the Raritan–Lower Bay complex by Ziskowski et al. (1980), who reported an average prevalence of 0.56% from a sample of 4,493 specimens. Our results (Table 8) indicate no demonstrable change in the prevalence of bentfin among stations or years but were considerably higher than those reported by Ziskowski et al (1980). The grouped station means of bentfin prevalence for the three time intervals were remarkably consistent at 2.09%:2.09%:1.34%. Our 1986–87 precessation prevalence estimates were not significantly different (P=0.6) from those of the postdumping period.

Discussion

Sindermann (1979) reviewed the history and associative evidence of pollution-related diseases and abnormalities of fish and shellfish. He found many of the effects leading to the correlation of diminished water quality with impaired health of fish to have been observed in artificial environments, such as in hatchery conditions where crowding enhances the growth and contagion of pathogens. Observations of fish disease also have been made in natural environments that have been altered artificially, such as in tidal waters influenced by heated effluents. Such effluents encourage local productivity, enhance growth of parasites and microorganisms, and serve to aggregate fishes during periods they would normally emigrate into deeper and more temperate waters.

Observations during this study were made in an essentially unrestricted environment; there were no physical or hydrographic barriers between study sites. The experimental approach to hypothesis building, in the sense of comparing control versus experimental sampling blocks, was confounded by several factors, however. One is the discharge plume of the Hudson–Raritan drainage, which sweeps out of the bay around Sandy Hook and progresses southward along the New Jersey shore. The input of waterborne contaminants into the 12-MDS area obviously continued beyond the termination of sludge dumping. A second obfuscation involves the interpretation of data from a migratory species (Phelan, 1995); the simplest migratory model has winter flounder moving from offshore into bays and estuarine portions of rivers from late December to early May for spawning. Young remain in the estuaries during their first year, and the older spawning segment of the stock exits the estuaries to oversummer in nearby ocean waters. This local population is in constant flux and no resident group is known to occupy a particular location for extended periods. Despite these potentially confounding factors, significant temporal trends were evident in the prevalence of some diseases (e.g. fin rot and internal lesions) which were contemporaneous with the cessation of sludge dumping. The prevalence rates of the various pathological conditions, albeit over a small area and of short duration, are unbiased, at least insofar as we know. The bimonthly sample series sustained for more than three years could be considered as simply a series of "snapshots" used for measuring pathological responses as an effect of sludge dumping. Together with other indicators of environmental quality, the continued monitoring of disease prevalence should provide evidence of the effects imposed by the cumulative presence of pollutants that alter the ecology of the New York Bight.

Summary

Although this study of winter flounder diseases in the New York Bight apex was of limited duration, some guarded conclusions are possible considering evidence from short-term trends and results of null hypotheses testing. The prevalence of fin rot and internal lesions apparently decreased. The monitoring history of fin rot, however, suggests that this condition was diminishing in winter flounder during intensive sludge dumping. No significant changes in prevalence of the internal parasite *Glugea stephani* or viral infections of lymphocystis could be related to cessation of sludge dumping. Similarly, there is no evidence to relate prevalence of somatic anomalies to the cessation of sludge dumping.

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Audience Questions

Question: Do you have any data on whether or not there were any changes in heterotrophic bacteria?

A. Pacheco: No, I do not, but Jay may.

J. O'Reilly: The slide I showed in the summary, from Linda Arlen and Randy Ferguson, showed two orders of magnitude decrease in total bacteria, the heterotrophic bacteria at the NY6 station. I also believe the fecal coliform densities parallel the decrease that I showed for the total count.

Question: Do you know what the genera were?

J. O'Reilly: No. No taxonomic workup was done, just total direct epifluorescent counts.

Question: It is a very interesting story, Tony. I think you convinced me certainly of the three declines that you are suggesting. I wonder if this, given the importance of fish disease as perceived by a lot of people, might not be something fairly important to look at again fairly intensively, say, in two or three years. Do you hope to do that?

A. Pacheco: You are talking to the right audience. I think we could find some interest to go back out and continue.

Effects of Sewage Sludge Disposal Cessation on Winter Flounder, Red Hake, and Lobster Feeding and Diets in the New York Bight Apex

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ABSTRACT

This paper presents results from a study of the response of feeding and diets of three common demersal species (winter flounder, *Pleuronectes americanus*; red hake, *Urophycis chuss*; and American lobster, *Homarus americanus*) to cessation of sewage sludge disposal in the coastal New York Bight after December 1987. Specimens of these species were collected bimonthly at three stations near the disposal site over 38 months, 1986–89. Results primarily from collections at the station nearest to the disposal site are used to examine response to cessation. In all three species, two feeding variables—percentage of empty stomachs and dominant prey in diets—showed little or variable quantitative change that could be related to cessation. Decreases in frequency of occurrence of certain artifacts and some minor prey species can be associated with cessation, however.

Introduction

In 1986 a study was begun to document expected environmental and biological changes resulting from cessation of sewage sludge disposal, planned for December 1987, in the New York Bight apex (EPD [Environmental Processes Division], 1988). Fish and lobster feeding and diets were monitored as part of this study because feeding inhibition has been cited as an effect of habitat degradation. Chemical contamination, hypoxia, or benthic macrofaunal community alterations, often associated with sludge disposal, are reported to inhibit feeding or alter diets (Fletcher et al., 1981; Spies, 1984; Collvin, 1985; Atchison et al., 1987; Bejda et al., 1987; Kramer, 1987; Weis and Khan, 1990). This monitoring information can be important to resource managers because chronically inhibited feeding, or diets based on low quality prey, can affect predator (i.e. fishery resource) growth, fecundity, or susceptibility to disease (Kurtz, 1975; Tyler and Dunn, 1976; Burton and Idler, 1987; Weis and Khan, 1991). This paper describes these monitoring results and identifies changes possibly associated with sludge disposal cessation, with a focus on the presumably most affected station, adjacent to the former disposal site.

Methods

Three sites in the New York Bight apex, R2, NY6, and NY11 (Fig. 1), with mostly similar habitat characteristics were selected for monitoring. Differences among the sites were related mostly to sediment organic carbon and chemical contaminant levels, development of seasonal hypoxia, and benthic macrofaunal community structure. These variables are thought to be associated closely with sludge disposal and were expected to change following cessation. Any changes in these variables were expected also to affect the trophodynamics of the benthic predators collected at the sites (EPD, 1988). Of the three study sites, station NY6 represented the most sludge-affected area and stations R2 and NY11 represented a gradient of sludge effects (see EPD, 1988, and Pikanowski, 1995, for details of overall study design).

Beginning July 1986, bimonthly trawl collections provided fish and certain megafaunal crustaceans for pathology and dietary analysis. Additional collections were made each August when bottom-water dissolved oxygen levels were generally lowest. From these collections three predators (red hake, *Urophycis chuss*; winter flounder, *Pleuronectes americanus*; and American lobster, *Homarus americanus*) were selected for trophodynamic analysis because of their availability, fishery value, and comparability to prior dietary studies in the area (Steimle, 1985; Steimle and Terranova, 1991). Samples were available from eight, 15-min trawls, each covering about 1 km, during daylight at each site for each bimonthly sampling. These trawls were distributed in an equidistant array intersecting or ending at the center of each site (see Pikanowski, 1995, for details). Bottom water, benthic macrofauna, and sediment samples were col-

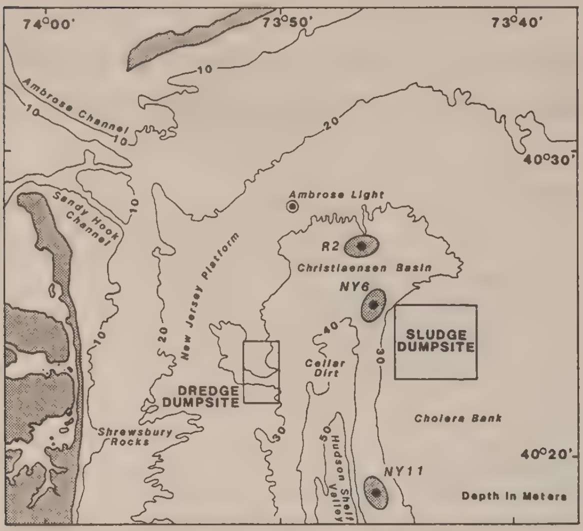

Figure 1

The locations of stations R2, NY6, and NY11 in the New York Bight apex, where fish and lobsters were collected for stomach content analysis.

lected before or after each trawl within the station ellipse (Fig. 1). Only two trawl collections were made at any site on the same day, morning and afternoon, to minimize possible influence of trawling at a site (e.g. exposed macrofauna) on trawl collection stomach contents.

Upon collection, winter flounder, red hake, and American lobster were removed and measured (total lengths). For most fish, the visceral cavity was quickly opened and the fish sexed. Exterior surfaces and viscera were scanned for obvious pathological conditions (see Pacheco and Rugg, 1995), and the stomach removed and its contents examined. Fish showing signs of regurgitation or an extruded gas bladder or stomach were not included in the stomach contents analysis. Small fish and thoraxes of all lobsters were preserved in 10% buffered formalin (with visceral cavity perforated to allow preservative penetration without damage to stomachs) for later measurement, sexing, and stomach removal for examination under magnification in the laboratory. The few recently molted (soft-, "leather"-, or "egg"-shelled) lobsters collected were not used for analysis because it is probable they would not be feeding outside their molt shelter and their stomach contents would not be representative of general site conditions (Scarratt, 1980). Lobster size was recorded as rostral-carapace length to be consistent with results of Wilk et al. (1995). Lobster eyesocket-carapace lengths were also recorded and a mean eyesocket to rostral-length ratio (1.355+/–0.027, 1 SD) was calculated from 100 random samples; this ratio was consistent within the 5.0–13.5-cm rostral-length range found in the collections. This ratio was used to estimate rostral-carapace lengths of lobsters with broken rostrums.

Stomach-content analysis was based on a semiquantitative method, standard in the Northeast Fisheries Science Center (Langton et al., 1980). This method involves emptying the stomach bolus into a petri dish, retaining the general shape of the bolus from the stomach, and estimating total stomach volume (to 0.1 cc) by a visual, side-by-side comparison of the bolus to the most appropriate of a series of volume-calibrated, variable diameter (0.5–2.5 cm) cylinders. Chyme from the stomach was not included in volume estimates. Boluses were then separated, and the contents were sorted and identified as far as possible. Percent contribution of each prey or stomach content item to the total stomach volume was also estimated visually. Bolus volumes were occasionally measured in water-filled graduated cylinders to check the accuracy of visual estimates of volume. These volumetric calibrations showed that visual estimates were reasonably reliable, i.e. within 10% of measured volumes. Each definable stomach item was counted and measured, and its degree of digestion noted. Most prey items were macroscopic and usually included larger, more common benthic species that are identified readily by eye or with a low power hand lens. Very small or questionable items found in the field examinations were preserved for later microscopic examination. Advantages and disadvantages of this method have been reviewed by Hyslop (1980) and Bowman[1] who found the method to provide reasonable results.

Other measures included percent-empty stomachs and a relative-fullness index (estimated total stomach volume divided by predator length) for evidence of feeding inhibition or change after disposal cessation. Percent frequency-of-occurrence analysis was used to define major components of the diet. This method has been recommended particularly for lobster (Elner and Campbell, 1987). Predator size is not discussed here as a variable because of the relatively small and irregular sample sizes for red hake and lobster and predominance of the 20 to 28-cm size class in winter flounder collections.

To test for significant changes in the percent-empty stomachs, relative fullness, or the contribution of certain "indicator" prey types or sludge artifacts before and after disposal cessation, data for each station were pooled into two temporal data sets. The first set included data from collections made between July 1986 and September 1987 and represent ongoing sludge disposal conditions. The second set included collections between July 1988 and September 1989, representing conditions after disposal cessation. Each of these data sets contained ten collection periods covering the same 15 collection-month distribution, from July to September, and excluded four transitional collections, made from November 1987 to May 1988 when disposal was almost phased out and immediately after abatement, when residual disposal effects would be expected to be nearly as important as while disposal was ongoing. This approach provided data sets that were expected to be large enough to estimate confidently the difference in the quantity of food in fish stomachs between locations or time periods, as suggested by Pennington et al. (1982). A 2 X 2 contingency table and *G* statistic (Windell and Bowen, 1978; Sokal and Rohlf, 1981; Crow, 1982) was used to test null hypotheses on the percent-empty stomach data sets. Statistical significance is $P=0.05$, unless stated otherwise.

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[1] Bowman, R. E. 1982. Preliminary evaluation of the results of analysis of the stomach contents of silver hake (*Merluccius bilinearis*) aboard ship and in the laboratory ashore. Northeast Fisheries Science Center, National Marine Fisheries Service, NOAA, Woods Hole, MA, Woods Hole Lab. Ref. Doc. 82-25, 13 p.

Results

Results of the stomach content examinations and associated sample population characteristics are presented separately for each of the three predators. Focus is on the NY6 collection site where an abatement change would be expected to be most apparent; some comparisons are made also with data from the other collection sites.

Winter Flounder

Winter flounder sample sizes varied; the cumulative number of samples per eight-trawl collection set at a station generally averaged >30 fish. Average size of flounder in samples ranged between 20 and 28 cm, but there were seasonal differences in the size range. Smaller fish (20–22 cm) dominated collections in warmer months, July through September, and larger fish (26–28 cm) were dominant in late fall to early winter collections. About 55–60% of the fish collected at all stations were females with little evidence of temporal variability in this proportion.

As expected, percent-empty stomachs and relative-fullness indices are generally inversely related (Fig. 2). Percent-empty stomachs usually peaked in warmer months, when the relative-fullness index was lowest (Fig. 2). Comparison of pre- and postcessation mean percent-empty stomachs showed statistically significant differences at all sites (Fig. 3). Station NY6 fish had an increase in mean percent-empty stomachs, from about 13% to 26%, while there were decreases at stations R2 and NY11.

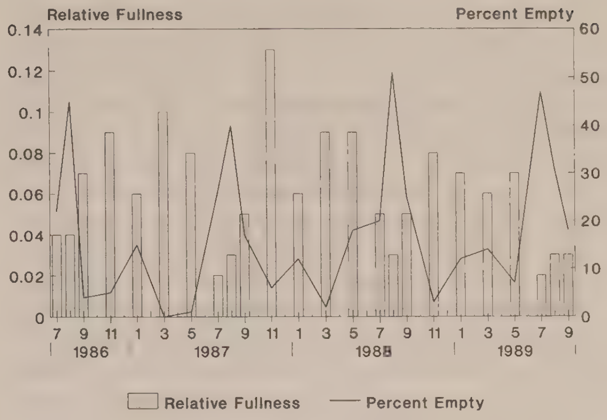

Figure 2
A summary of temporal trends in stomach relative-fullness index and percent empty stomach values of winter flounder, *Pleuronectes americanus*, from station NY6 in the New York Bight apex. Sewage sludge disposal ceased in December 1987.

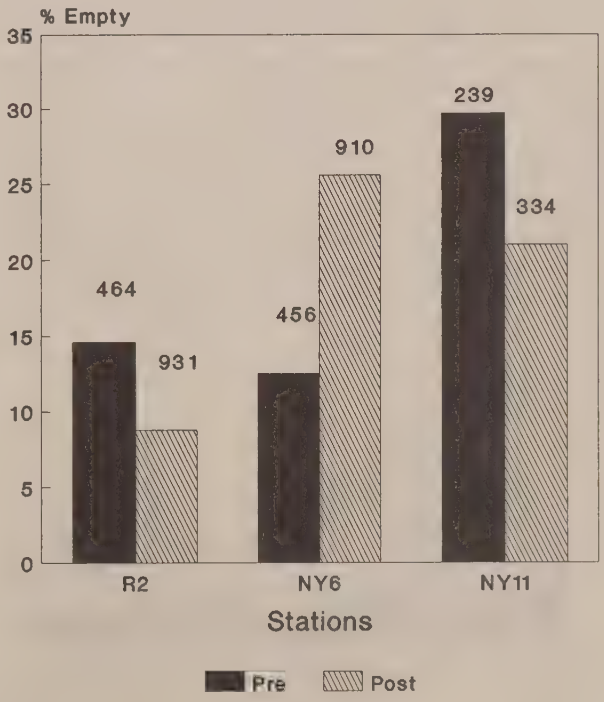

Figure 3
A comparison of percent empty stomachs in winter flounder, *Pleuronectes americanus*, at the three replicate stations in the New York Bight apex during pre– and post–sewage sludge disposal cessation periods. Period of pre–sewage sludge disposal=July 1986–Sept. 1987; post-sewage sludge disposal=July 1988–Sept. 1989. Value above each column is the sample size.

A few prey taxa dominated the diets at all stations but there were differences among stations in the proportional contribution of some prey species to the diet. Overall, winter flounder diets were dominated by the tube-dwelling anemone *Ceriantheopsis americanus*, a rhynchocoel, undoubtedly *Cerebratulus* sp., several polychaete species, and a few small crustaceans, e.g. the isopod *Edotea triloba*, and amphipods. There were temporal changes or trends in the contribution of certain prey to the diets, however. For example, there was an proportional increase in the contribution of the amphipod *Unciola* sp. to the diet at station NY6 (Fig. 4). The mean frequency of occurrence of capitellid polychaetes showed substantial decreases in the pre- and postcessation comparison (Fig. 5).

Red Hake

Red hake were more irregular in collection abundance than winter flounder and seldom resulted in cumulative samples of 30 fish per eight-trawl collection set. There was also wider variability (0–100%) in the proportion of females in the collections. The lengths of red hake ranged from 10 to 34 cm; but most were between 17 and 30 cm. There was little evidence of seasonal trends in relative abundance or in the sex or size composition of the red hake.

Percent-empty stomach and relative-fullness indices for red hake were variable and without clear seasonal or other temporal trends. The results suggested, however, that more empty stomachs and lower relative-fullness values occurred in mid-summer collections, although sample sizes were smaller (Fig. 6). Comparing pre- and

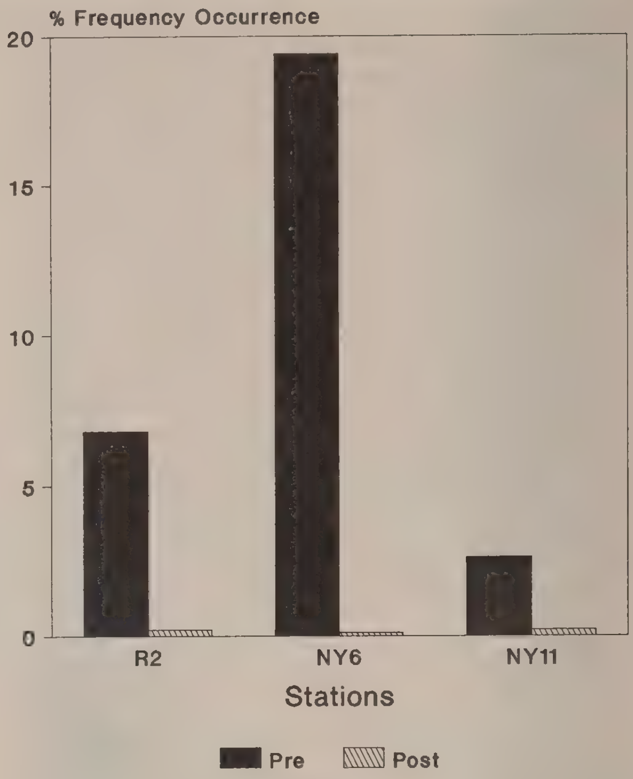

Figure 5
A comparison of the percent frequency occurrence of the stress-indicator polychaete *Capitella* sp. in the diet of winter flounder, *Pleuronectes americanus*, from three stations in the New York Bight apex (see Figure 3 for winter flounder sample sizes) during pre– and post– sewage sludge disposal cessation periods. Period of pre–sewage sludge disposal=July 1986–Sept. 1987; post-sewage sludge disposal=July 1988–Sept. 1989. Sewage sludge disposal ceased in December 1987.

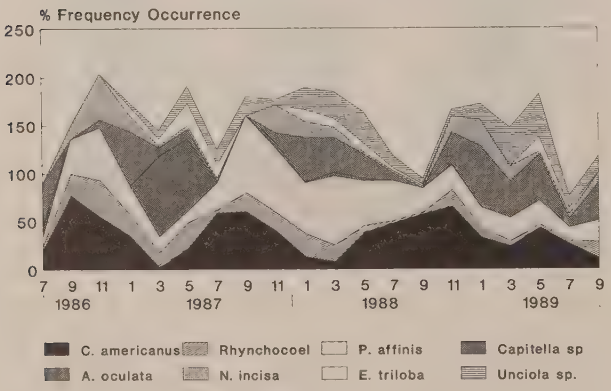

Figure 4
A summary of temporal trends in major prey by percent frequency of occurrence (frequency of occurrences overlap and thus exceed 100%) consumed by winter flounder, *Pleuronectes americanus*, at station NY6 in the New York Bight apex. Sewage sludge disposal ceased in December 1987.

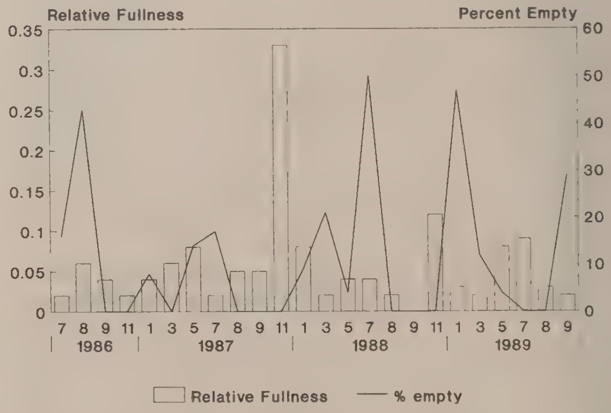

Figure 6
A summary of temporal trends in stomach relative-fullness index and percent-empty stomach values of red hake, *Urophycis chuss*, from station NY6 in the New York Bight apex. Sewage sludge disposal ceased in December 1987.

postcessation periods showed that only the percent-empty stomach increase at station NY6 was statistically significant; slight increases, not significant, were noted at the other stations.

Dominant prey were similar for all stations and consisted of three species: the polychaete *Pherusa affinis* and two decapod shrimp, *Crangon septemspinosus* and *Dichelopandalus leptocerus* (Fig. 7). Atlantic rock crabs *Cancer irroratus*, the polychaete *Nephtys incisa*, and fish remains were also important food at station NY6. Stomachs occasionally contained small quantities of artifacts: small pieces of sheet plastic, coal and coal ash, paint chips, thin rubber (latex?) sheets, "band aids," cloth, metal foil, foil and plastic pill packaging, and synthetic twine (in approximate order of frequency of occurrence), and other non-food items such as hair, fibers, pieces of wood, twigs, and a feather. Overall, their frequency-of-occurrence showed no clear relation to cessation of disposal, except for reduced occurrence of hair and fibers.

American Lobster

Lobster sample sizes were usually small (<20 individuals per collection period) with some collection periods not yielding samples. There was substantial variability in sex ratios among collections. A seasonal cycle in abundance, well known to the fishery, was apparent with minimal availability in the winter months, especially March. Most lobsters collected were between 8 and 10 cm in rostral-carapace length (6–7.5 cm eyesocket-carapace length).

Percent-empty stomach and relative-fullness index results suggested no consistent temporal patterns or trends. There was evidence at all stations, for example, of the relative-fullness indices being highest during the summer and fall, but percent-empty stomachs were often elevated at about the same time (e.g. see Fig. 8). Pre- and postcessation comparisons resulted in slight, but not statistically significant, increases in percent-empty stomachs after cessation.

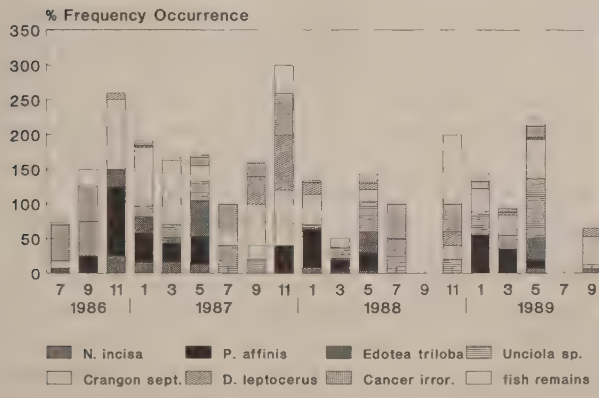

Figure 7

A summary of temporal trends in major prey, by percent frequency of occurrence (frequency of occurrences overlap and thus exceed 100%), consumed by red hake, *Urophycis chuss*, at station NY6 in the New York Bight apex. Sewage sludge disposal ceased in December 1987.

Lobster diets had considerable overlap of dominant prey among stations. Fish, Atlantic rock crabs, *C. irroratus*, crab fragments (mostly *C. irroratus*), hair, and fibers (primarily fine, colored, synthetic threads) were found frequently in stomachs at all stations, e.g. NY6 (Fig. 9). The frequency of occurrence of hair and fiber (presumably constituents of sewage sludge) declined substantially at all stations after cessation (Fig. 10). The polychaete *P. affinis* was also common in the lobster diet (Fig. 9). Also, after 1987 there was a decline in the contribution of the small nut clam, *Nucula* sp. (Fig. 9). Besides hair and fibers, lobster stomachs also occasionally contained other artifacts: sponge-like pieces of rubber, rubber bands, small pieces of hard plastic, polystyrene-like pellets, pieces of thin plastic sheets, metal foil, synthetic twine, paint chips, coal or coal ash, and pieces of wood or twigs. These occurrences were too few and random to support any trend or reveal any cessation-related differences.

Discussion

The hypothesis that cessation of sewage sludge disposal at the 12-mile sewage sludge dumpsite would (or would not, as a null hypothesis) cause a change, specifically an improvement, in any of the feeding and diet variables

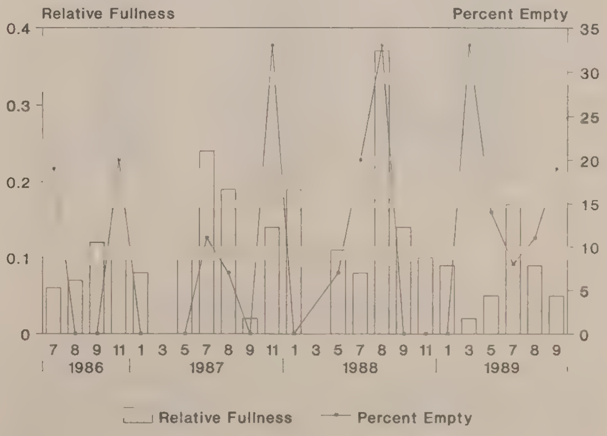

Figure 8

A summary of temporal trends in American lobster, *Homarus americanus*, stomach relative-fullness and percent empty stomachs values from station NY6 in the New York Bight apex. Sewage sludge disposal ceased in December 1987.

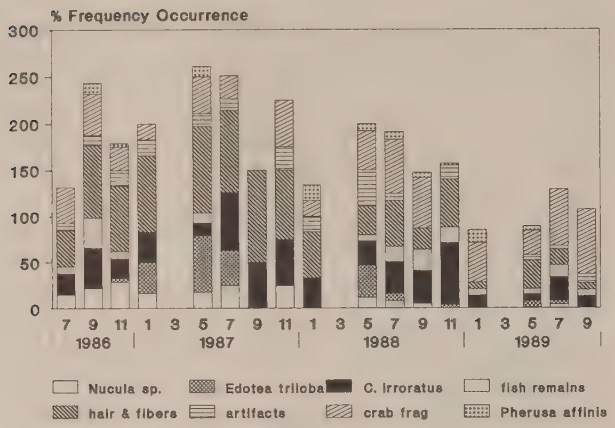

Figure 9
A summary of temporal trends in major prey, by percent frequency of occurrence (frequency of occurrences overlap and thus exceed 100%), consumed by American lobster, *Homarus americanus*, at station NY6 in the New York Bight apex. Sewage sludge disposal ceased in December 1987.

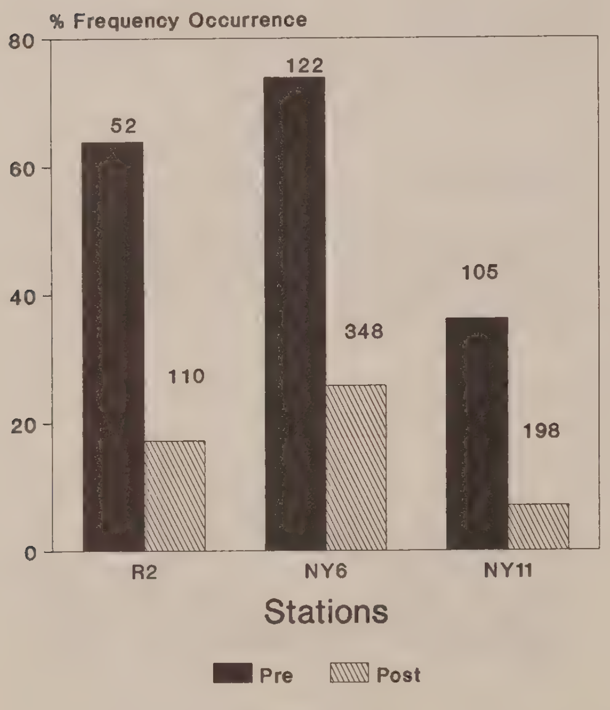

Figure 10
A comparison of percent frequency of occurrence of hair and textile fibers in the stomachs of American lobster, *Homarus americanus*, during pre– and post–sewage sludge disposal at three sites in the New York Bight apex; value above each column is the sample size. Period of pre–sewage sludge disposal=July 1986–Sept. 1987; post-sewage sludge disposal=July 1988–Sept. 1989. Sewage sludge disposal ceased in December 1987.

examined for three demersal predators at the three study sites is variably supported by the results.

Percent-empty stomach and relative-fullness results, for example, provide mixed evidence to accept or reject the null hypotheses concerning a temporal change or improvement after cessation at the study sites. For winter flounder, the percent-empty stomach means were significantly lower at stations R2 and NY11 after cessation (Fig. 3). This would be an expected improvement response, assuming some sludge disposal factor was inhibiting feeding. In contrast, the pooled mean of percent-empty stomachs was statistically significant, and two times higher at station NY6 (Fig. 3), suggesting a continued sludge or other undefined effect at that site.

Less significant results were found for red hake, although these were frequently based on small sample sizes or there were collection gaps. Red hake percent-empty stomachs for all stations showed an increase after cessation, but this change was statistically significant only for station NY6. Similar results were found for lobster with no statistically significant change or improvement in feeding at any of the study sites, although percent-empty stomachs increased to some degree at all stations. Postcessation increases in mean percent-empty stomachs for winter flounder at station NY6 (Fig. 3) and red hake and lobster at all stations, although not always statistically significant, contradicts the expected response of a decrease in percent-empty stomachs to the decrease in some sludge-related conditions, e.g. chemical contaminants (Zdanowicz et al., 1995), that could inhibit feeding. For red hake and lobster that feed heavily on Atlantic rock crabs, this increase in empty stomachs could be a response to the significant decline in abundance of Atlantic rock crabs found in the area after cessation (Pikanowski, 1992). For winter flounder, which feed mostly on smaller, surface-dwelling benthic prey, the significant increase in percent-empty stomachs at NY6 could be a response to the decrease in the mean total biomass (from 6343 to 2836 mg/0.1 m^2) of benthic macrofauna after abatement at that site; macrofaunal biomass decreases were not evident at other stations (Reid et al., 1995).

Changes in major components of the diets of the three selected predators do not support a substantial effect of disposal or a change because of cessation. The major prey of winter flounder at all stations did not change drastically after cessation (Fig. 4) or differ substantially from that previously reported for the area or in comparable studies for the Middle Atlantic Bight (Steimle and Terranova, 1991). More pollution-tolerant species, e.g. capitellid polychaetes (Fig. 5), were consumed during the disposal period, however, which corresponds to their high relative abundance during this period (Reid et al., 1995). The consumption of amphipods, generally considered a pollution-sensitive taxon, did increase after abatement, however. Amphipod and capitellid abundances are often used as indicators of habitat quality, and this response is most likely reflected in their changing contribution to diets.

Major components of the diet of red hake, i.e. large polychaetes, decapod crustaceans, and fish, also showed little change after abatement (Fig. 7) or were not substantially different from other studies of red hake diet (Steimle and Terranova, 1991). Feeding by red hake is thought to be discontinuous and could be less controlled by daylight than is winter flounder feeding. Red hake can detect prey tactually and chemoreceptively, as well as visually (Luczkovich and Olla, 1983).

Diets of lobster in this study were dominated by decapod crustaceans, especially *C. irroratus* (Fig. 9). This is similar to results from other lobster feeding studies, allowing for some differences expected for different zoogeographic zones and habitat types (Weiss, 1970; Ennis, 1973; Scarratt, 1980; Elner and Campbell, 1987; Karnofsky et al., 1989). The presence of artifacts in the lobster stomachs is not unusual; rubber, metal, rope, plastic, wood, plastic spheres, nylon, tea bags, and other "non-marine items" have been reported in other lobster stomach-content studies (Weiss, 1970; Ennis, 1973; Scarratt, 1980; Elner and Campbell, 1987; Hudon and Lamarche, 1989). The hair and fibers common in the New York Bight apex lobster stomachs (Figs. 9, 10), however, have not been noted before in the literature. Their presence and decline in the stomachs (Fig. 10) would appear to be related to sewage sludge disposal and its cessation. The significance to lobster of the hair and fibers is unknown; but significant quantities could possibly interfere with digestion, stomach evacuation rates, or stomach lining molting.

The results of this study are subject to the biases and potential errors common to dietary studies of this type (Bowman[2]). For example, the stomach content results are undoubtedly biased towards larger, more slowly digested items or calcified remains (Carter and Steele, 1982; MacDonald et al., 1982) and influenced by temperature or other factors.

Besides these potential biases, it is possible that the stomach contents may represent some feeding outside the 0.5–0.75 km diameter station trawling ellipse (Fig. 1) because of predator mobility. For example, Lund et al.[3] reported mean daily lobster movement in Long Island Sound to be less than 0.5 km, based on ultrasonic tag monitoring, but this was primarily at night when most feeding is reported to occur (Scarratt, 1980). Thus, some of the lobsters collected at a station could have fed outside the station ellipses before moving to the station area. This possibility would be greater for winter flounder, which have been reported to "shamble" across the bottom at an average speed slightly exceeding 1 km/hr (MacDonald, 1983). Thus, they could be active and feeding for several hours before late morning collections. The swimming speed and daily foraging range of red hake may be greater, with most of their movement and feeding, like lobster, also being at night (Bejda[4]). If there are no sharp environmental or forage gradients near any study site, feeding near but outside the station ellipse would not lead to erroneous conclusions about site-associated feeding.

The general lack of a significant or consistent change in trophodynamic indicators, such as percent-empty stomachs, of the predators in this study will not necessarily prove the absence of a significant trophodynamic-related effect from waste disposal. Feeding can be maintained or even increased during stress to compensate for increased energetic losses caused by pathological or physiological impairment, e.g. elevated gill ventilation rates or poor food metabolism (Collvin, 1985). There could be qualitative effects, also, such as the consumption of chemically contaminated prey to be considered (Spies, 1984; Steimle et al., 1994).

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Audience Questions

Question: Since lobsters are migratory, might they simply be moving through the area when you are collecting them? Was there any work done or any indication that some of those remained in that particular area?

F. Steimle: No tagging studies were done with lobsters; that would be the only way to get at that question. If the "hairballs" reflect a sewage sludge effect, then perhaps if somebody were looking at lobsters outside the area and found hair in the stomachs, this could suggest where the lobsters were going and be used as an indicator of migration. I do not think that has been done. In all the lobster diet studies that I have read, I have not seen any note of this type of material—hair or fibers—in the diets. Although they have noted wood, pebbles, and old tea bags among a number of different things that lobsters would eat, hair was not noted.

Question: The same question could be asked of winter flounder. They could simply be moving through the area; and when you are intercepting them they may be moving to or from a more desirable area.

F. Steimle: Yes, that is what I meant by the mobility question. As Beth [Phelan] mentioned, fish are moving all over the place, and that is the problem with trying to use a mobile sampler to sample a specific site. We can only assume that the fish were probably feeding pretty close to the area where we collected them. That is what I meant by the stomach contents representing an areal estimate. There is some evidence that suggests, for example, that winter flounder can swim up to about a knot and a half. By the time we collect them in midmorning, it is possible that the fish could have moved five miles between when they started feeding and when we collected them, if they were moving in a straight line.

Red hake could be the same or even greater in terms of their motility. They tend to move more at night; lobsters, less. There is some evidence to suggest that maybe they could move perhaps a half a kilometer during the night in some of their feeding forays. This is not a good approach to a site-specific study because we do not know precisely where the feeding was going on. We have to presume that a majority of the feeding and the gut contents came from close to the area where we collected them.

Question: Will you make any comment on how the population density in New York–New Jersey area relates to hairballs in the diets of those animals?

F. Steimle: The only thing I can say is that the sediment study group has reported seeing some hair in the sediment samples that they are analyzing. I did not have the hair analyzed to determine what type of hair it was. I can say it was black, stiff, and looked like mammalian hair of some sort. The other fibers were colored in many cases and looked like sweepings from some textile house or similar industry—different colors, different length, both straight and crinkly.

Question: I wondered if you might speculate, or if you have any thoughts, on what impacts the mud dumpsite [dredge spoil site] might have on your results or be confounding the studies that you and Tony Pacheco have done?

F. Steimle: All of our collections were to the east, from the three replicate sites. I really can not speculate. Several years ago I did some diet analyses of fish collected in that area (Steimle, F., and R. Terranova, 1991 [Trophodynamics of select demersal fishes in the New York Bight. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-F/NEC 84, 11 p.]). We found no significant difference in what they fed on at the west side of the mud dumpsite as opposed to the east side, where the sewage sludge dumpsite is. But we have not specifically focused or done a lot of work with the dredge spoil or mud dump.

Limited Responses of Benthic Macrofauna and Selected Sewage Sludge Components to Phaseout of Sludge Disposal in the Inner New York Bight*

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ABSTRACT

Information on responses of marine ecosystems to waste disposal, and to abatement of disposal, is needed to manage wastes properly and to conserve and manage living marine resources. For this reason, a multidisciplinary study was conducted of ecosystem responses to phaseout of sewage sludge disposal in the inner New York Bight. Sludge dumping at the 12-Mile Sewage Sludge Dumpsite, 22 km off northern New Jersey, began in 1924 and was phased out from March 1986 through December 1987. The study of responses involved monthly sampling of numerous water, sediment, and biological variables from July 1986 through September 1989. Included were triplicate samples for analysis of benthic macrofauna and selected sludge components at three stations chosen to represent 1) greatest sludge accumulation and effects, 2) little or no sludge influence, and 3) an intermediate situation with organic enrichment perhaps partly due to sludge. In the most sludge-altered area, some benthic variables showed clear responses to the phaseout, while others did not. Numbers of overall benthic species, as well as numbers of species of mollusks, crustaceans, and amphipods, increased significantly relative to numbers at the other two stations. There were significant decreases in abundances of *Capitella* spp. and nemerteans, taxa which had dominated the fauna of the altered area during dumping. Conversely, abundances of

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ABSTRACT (continued)

amphipods (especially *Photis pollex* and *Unciola irrorata*), which had been rare or absent in the altered area, increased substantially. However, there was little evidence of colonization of the altered area by species that were dominants at the other stations. Cluster analysis of benthic species composition also indicated changes in the fauna of the altered area after cessation but little increase in resemblance to the other stations. Continued high abundance of tomato seeds and other detritus apparently of sludge origin in the altered area indicated the persistence of some sludge components. When the stations were resampled 34 and 39 months after cessation, seeds were still prominent in the altered area, while quantities of other sludge detritus appeared to have decreased. In terms of numbers of species and abundance of *Capitella* spp. and nemerteans, this area could be considered to have responded strongly to removal of sludge influence by 39 months after cessation, but there was still little colonization by dominants from the other stations.

Introduction

Ocean disposal of sewage has been a fairly common practice, and considerable effort has gone toward determining fates and effects of the sewage (Gross, 1976a and 1976b; Geyer, 1981; Dart and Jenkins, 1982; Mayer,1982; Spies, 1984; Brooks et al., 1987; Bascom, 1989; Norton, 1989; Ferraro et al., 1991). Many studies of biological effects of sewage disposal have concentrated on benthic macrofauna (usually defined as the bottom-living invertebrates collected in grab samplers and retained on 0.5–1.0-mm mesh sieves). Due to factors such as their limited mobility and wide range of documented responses to pollution, benthic macrofauna are considered among the best groups for monitoring biological effects of sewage disposal and of pollution in general (Boesch, 1974; Swartz, 1978; Gray, 1980; Kuiper, 1986; Bilyard, 1987; Smith et al., 1988; Jackson and Resh, 1989). Benthic macrofauna are also important as a food and contaminant source for bottom-feeding fishes and crustaceans of direct importance to man (e.g. Boesch, 1982; Steimle, 1985; Steimle and Terranova, 1991).

As part of a multidisciplinary study of the effects of dumping sewage sludge on habitats and biota of the inner New York Bight (EPD [Environmental Processes Division], 1988), this paper concentrates on effects on benthic macrofauna. Effects of sewage sludge dumping on benthic macrofauna have been studied extensively in the Bight (Rowe, 1971; Pearce et al., 1981; Boesch, 1982; Reid et al., 1982a and 1982b; Steimle et al., 1982: Steimle, 1985) and around dumpsites off Delaware (Watling et al., 1974; Devine and Simpson, 1985), England (Eagle et al., 1978; Norton and Champ, 1989), Scotland (Mackay, 1986; Pearson, 1987; Moore and Rodger, 1991), and Germany (Caspers, 1987). Findings of the present study can also be compared with effects of other types of sewage disposal, e.g. pipeline discharges of sewage sludge (Southern California Coastal Water Research Project[1,2]) and sewage wastewater (Stull et al., 1986a and 1986b; Swartz et al., 1986; Maurer and Haydock, 1989; Ferraro et al., 1991) off southern California. Responses of benthic macrofauna to the phaseout of dumping in the inner New York Bight may help in evaluating responses to reduction of sewage inputs to Boston Harbor and other temperate coastal habitats. Results from this study may also aid in understanding and predicting effects of other organic inputs, such as from fish farming, and perhaps other ocean dumping (e.g. dredged material) and ocean waste disposal in general.

In the New York Bight, determination of sewage sludge effects on benthic macrofauna has been confounded by other waste inputs to the area. Sludge disposal ranked only third in loadings of organic carbon and most other contaminants to the inner Bight, with dredged material first and the Hudson-Raritan outflow second (Stanford and Young, 1988). The phaseout of sewage sludge dumping in the inner Bight between March 1986 and December 1987 offered an opportunity to examine responses of benthic macrofauna, and thus to infer the effects of sludge dumping and to document rates of responses seen following the phaseout.

Materials and Methods

Study Area and Sewage Sludge Inputs

The inner Bight sludge dumpsite, known as the 12-Mile Site, is located 22.2 km east of Sandy Hook, N.J. (Fig. 1).

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[1] Southern California Coastal Water Research Project (SCCWRP). 1986. Biennial Report for the years 1985–1986. SCCWRP, W. Pacific Coast Highway, Long Beach, CA 90806.

[2] SCCWRP. 1989. Biennial Report for the years 1987–1988. SCCWRP, W. Pacific Coast Highway, Long Beach, CA 90806.

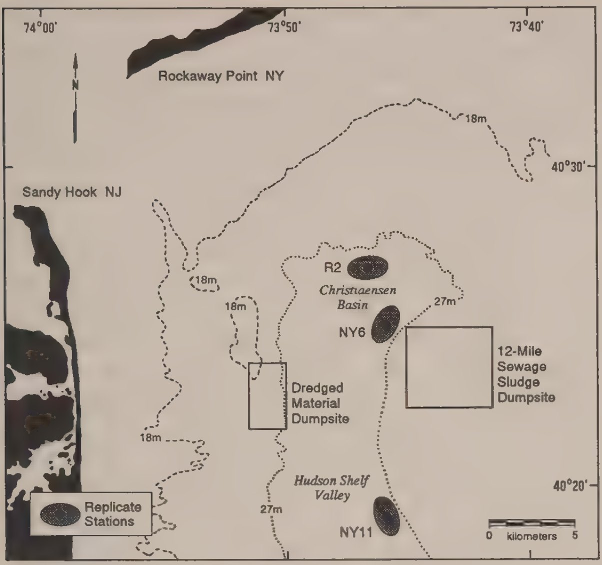

Figure 1

Locations of dumpsites and stations in the inner New York Bight.

Dumping at the site began in 1924 (EPD, 1988). Since 1960, when dumping rates were recorded, there was a general increase, to a maximum of 7.6 million t (wet weight) in 1983 (Suskowski and Santoro, 1986). Sludge inputs in the early 1980's were, at the time, the largest known to any oceanic sludge dumpsite (Norton and Champ, 1989). Composition of the sludge changed over time as improved sewage treatment and lowered industrial inputs reduced contaminant concentrations. A comparison of 1973 and 1987 loadings (Hydroqual[3]) indicated decreases over that period, some quite large, in total sludge solids, biochemical oxygen demand, and most heavy metals, e.g. loadings of cadmium, chromium, and mercury were reduced by at least 45% (no 1973 data on organic contaminants were available for comparison).

The sludge dumpsite is located in 23.8 to 25.3 m water depth. During the period of dumping, dumpsite sediments contained slightly elevated concentrations of carbon and contaminants, but there was no long-term buildup of sludge materials at the site (Norton and Champ, 1989). Contaminant accumulation and effects were most apparent in the deeper waters (30 to 40 m) of the Christiaensen Basin to the west (Fig. 1), especially just west of the dumpsite's northwest corner, where most dumping occurred (EPD, 1988). The Christiaensen Basin in general contains finer surface sediments than the surrounding shallower sandy areas (Swift et al., 1976). Sediment carbon, trace metal and organic contaminant concentrations, and biomass of benthic macrofauna, are elevated in the Basin compared to the rest of the Bight (Pearce et al., 1981; Boesch, 1982; Reid et al., 1982a and 1982b; Steimle et al., 1982; Steimle, 1985). The Basin is the head of the Hudson Shelf Valley, which extends from the inner Bight southeast to the Hudson Canyon at the continental shelf edge. Like the Basin, the Shelf Valley is deeper and has finer sediments than most of the surrounding shelf (Swift et al., 1976).

Sampling Design

Between July 1986 and September 1989, the National Marine Fisheries Service and collaborators conducted monthly surveys for responses of habitats and biota of the inner Bight to the phaseout of sludge disposal. Sampling included numerous water, sediment and biological variables. Further description of the study plan is given in EPD (1988) and Pikanowski (1995).

Benthic sampling consisted of three samples taken monthly at each of three "replicate" stations (Fig. 1). Station NY6 was located at a depth of 31 m, 1.6 km west of the dumpsite's northwest corner, in the area of greatest apparent sludge buildup and effects (EPD, 1988). Station NY11 was in 29 m of water, 11.3 km south of NY6 on the eastern shoulder of the Shelf Valley. It was chosen as a reference area since its depth and sediment type were similar to those at NY6, but low levels of sediment contaminants indicated minimal sludge influence (EPD, 1988). Station R2 was at 28.5 m depth in the northern Christiaensen Basin, 3.4 km north of NY6. It was considered intermediate between NY6 and NY11 in sludge influence, with somewhat elevated concentrations of sediment contaminants and a benthic faunal biomass apparently enriched by organic carbon inputs from sludge and other sources (EPD, 1988).

Samples were also taken bimonthly at five locations on an ellipse around each station (Fig. 1) and at 22 other stations of a "broadscale" survey covering most of the inner Bight (EPD, 1988; Pikanowski, 1995). Analysis of ellipse and broadscale benthic samples has not been completed, but data are available for many other aspects of the overall study.

Bottom samples were taken with a 0.1-m^2 Smith-McIntyre grab. After vertical profiles of sediment redox potential were measured and subsamples were taken for sediment grain size, trace metals, organic contaminants, organic carbon, and other variables (Draxler, 1995), the remainder of each grab sample was rinsed on a 0.5-mm sieve. Retained materials were fixed in

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[3] Hydroqual. 1989. Assessment of pollutant inputs to New York Bight. Report to U.S. Environmental Protection Agency from Hydroqual, Inc., 1 Lethbridge Plaza, Mahwah, NJ. Unpubl. manuscr., 117 p.

10% buffered formalin with Rose Bengal. One to three days later the samples were transferred to 70% ethanol with 5% glycerin. Dissecting microscopes were used to sort organisms from sediment. Identifications were to species level whenever possible, except for Nemertea. Oligochaetes, archiannelids, and colonial forms (e.g. bryozoans, tunicates and sponges) were not enumerated due to our uncertainty of identification and/or difficulty of quantification. Tomato seeds, a potential sludge tracer (Pearce, 1971), were also counted in most samples. Wet weight macrofauna biomasses were determined by blot-drying each taxon on absorbent paper toweling for 3 min and weighing to the nearest mg on an electronic balance. Biomass data are presented with and without weights of two bivalves, *Arctica islandica* (ocean quahog) and *Pitar morrhuanus* (false quahog), whose occasional high values greatly increased biomass variability.

Sampling and data analysis reflect the fact that with only one dumpsite being phased out, there can be no true replication in the classic experimental sense. Stewart-Oaten et al. (1986) suggest that in such situations, valid impact assessments may still be achieved by using "pseudoreplication in time," i.e. taking samples over time both before and after the impact (or cessation, in this case) at both control and impacted sites (the BACI design). Differences between the "before" data were statistically compared with "after" differences to determine if the two periods were significantly different. Both parametric (Student's t) and nonparametric (Mann-Whitney U) tests of differences were used. Data were also tested both parametrically (C test) and nonparametrically (runs test) (Zar, 1984) for serial correlation, since tests of differences may not be valid if data are serially correlated. No serial correlation that would require qualifying the results was found (see Pikanowski, 1995, for further discussion of these analyses).

At the study's outset, several anticipated responses of the macrofauna to phaseout were stated as hypotheses: 1) there would be significant increases in the total numbers of species, in crustacean and amphipod species, and in amphipod individuals at stations NY6 and R2 as compared to NY11, and 2) there would be a significant decrease in mean densities of *Capitella* spp. at NY6 (EPD, 1988). Other variables analysed for differences using BACI were numbers of species of mollusks, numbers of individuals of the amphipod *Unciola irrorata*, numbers of tomato seeds, overall biomass, "adjusted" overall biomass (without the large bivalves *Arctica islandica* and *Pitar morrhuanus*), biomass of nemerteans, and biomass of the polychaete, *Pherusa affinis*.

Q-mode cluster analysis (clustering stations according to abundances of species they have in common) (Boesch, 1977; Romesburg, 1984) was performed using means of the abundances of all commonly occurring species for each station and month. To facilitate computation, the analysis used only the 150 most frequently occurring species (based on percent occurrence in station/month combinations and in individual samples, plus percent of grabs with >10 individuals of the species in question). All data were transformed (ln [count + 1]) before clustering, because of the presence of zero scores and to reduce the influence of very abundant species on inter-sample similarities (Boesch, 1977). Percent faunal similarities between stations were measured using the Bray-Curtis (1957) coefficient $Cz = 2w/(a+b)$, where a is the sum of abundances of all species found in a given sample, b is the sum of species abundances for another sample, and w is the sum of the lower of the abundance values for each species common to both samples. Clustering was performed using flexible sorting with the cluster intensity coefficient $\beta = -0.25$ (Boesch 1977). A similarity level of $Cz = 0.60$ was used to form subgroups from the overall dendrogram. This cutpoint was subjectively chosen because it generated a reasonable number of subgroups per station (nine for NY6 and four each for NY11 and R2), permitting analysis of temporal trends within stations. The strategy of Romesburg (1984) for objectively choosing a cutpoint could not be used, because the dendrogram had no areas above 0.2 similarity where the number of clusters remained the same over a wide range of similarities.

Semiquantitative observations were made on each grab sample as soon as it was brought aboard. Sediment type was noted, and amount of sludge-like detritus was estimated. Most of this detritus was probably cellulose fibers and plant fragments (Campbell et al., 1988); tomato and other seeds were also abundant.

The three stations were sampled again in October 1990 and March 1991. Field observations and data from these collections are included to extend the study's temporal coverage.

Results

The BACI analyses of selected faunal variables (Table 1) indicated several significant ($P<0.05$) responses of NY6 to phaseout of sludge dumping. Mean numbers of species per 0.1 m^2 (Fig. 2) were distinct between stations at the study's outset, in the order NY11>R2>NY6 as expected based on numbers of species typically found in these three areas in earlier studies (Reid et al., 1982b, 1991). Numbers of species increased at all stations, but numbers at NY6 increased significantly more relative to both NY11 and R2 in the before–after comparison (Table 1). Numbers of species at R2 did not increase relative to NY11, so the hypothesis of significant increases at both NY6 and R2 was only partly confirmed. Similarly, the hypotheses of increases in numbers of species of crustaceans (Fig. 3), numbers of species of

Table 1

Mean values for grab samples at replicate stations, per $0.1m^2$, before and after the cessation of sewage sludge dumping. Values labeled contrast are differences between the before and after periods. Negative values for **t** indicate increases at NY6 after cessation. The comparisons; R2-NY11, NY6-NY11 and NY6-R2 represent the technique of replication over time. The critical values for the Student's t-test and the Mann-Whitney **U** test are respectively, 2.026 and 259.

Group		Mean (Std. Error)								
		Before (18 Mo)		After (21 Mo)		Contrast (Before - After)		$t_{(2)}$ (d*f*=37)	$U_{0.05(2), 18,21}$	Significance (P = 0.05)
Total Species	NY6	19.17	(1.46)	33.02	(1.96)	–13.85	(2.51)	–5.51	344.0	+
	R2-NY11	–11.15	(1.13)	–9.52	(1.10)	–1.63	(1.58)	–1.03	222.0	
	NY6-NY11	–22.00	(1.22)	–12.96	(1.29)	–9.04	(1.79)	–5.04	335.0	+
	NY6-R2	–10.85	(1.07)	–3.44	(1.15)	–7.41	(1.59)	–4.66	324.5	+
Mollusk Species	NY6	2.98	(0.34)	5.86	(0.55)	–2.88	(0.67)	–4.28	320.5	+
	R2-NY11	–1.02	(0.43)	0.60	(0.40)	–1.61	(0.59)	–2.74	276.0	+
	NY6-NY11	–4.67	(0.40)	–1.29	(0.44)	–3.37	(0.60)	–5.59	341.0	+
	NY6-R2	–3.65	(0.29)	–1.89	(0.42)	–1.76	(0.53)	–3.33	295.0	+
Crustacean Species	NY6	2.61	(0.34)	6.89	(0.52)	–4.28	(0.65)	–6.63	370.0	+
	R2-NY11	–2.06	(0.36)	–2.26	(0.32)	0.21	(0.49)	0.42	201.0	
	NY6-NY11	–2.96	(0.41)	–1.06	(0.39)	–1.91	(0.57)	–3.34	299.5	+
	NY6-R2	–0.91	(0.29)	1.21	(0.29)	–2.11	(0.42)	–5.06	333.5	+
Amphipod Species	NY6	1.04	(0.20)	4.51	(0.45)	–3.47	(0.52)	–6.64	368.0	+
	R2-NY11	–1.68	(0.21)	–1.80	(0.30)	0.13	(0.37)	0.34	199.5	
	NY6-NY11	–2.27	(0.25)	–0.90	(0.32)	–1.37	(0.41)	–3.33	297.0	+
	NY6-R2	–0.59	(0.21)	0.90	(0.18)	–1.50	(0.28)	–5.43	336.5	+
Amphipod Individuals	NY6	3.22	(0.99)	91.21	(28.11)	–87.98	(30.43)	–2.89	360.5	+
	R2-NY11	–33.77	(9.32)	–14.06	(6.87)	–19.71	(11.38)	–1.73	231.0	
	NY6-NY11	–47.51	(9.11)	40.22	(23.25)	–87.73	(26.53)	–3.31	328.0	+
	NY6-R2	–13.74	(2.37)	54.29	(24.06)	–68.03	(26.13)	–2.60	296.5	+
U. irrorata Individuals	NY6	1.48	(0.46)	17.86	(3.68)	–16.38	(4.00)	–4.09	357.5	+
	R2-NY11	–11.71	(2.20)	–15.64	(3.24)	3.93	(4.05)	0.97	208.0	
	NY6-NY11	–10.77	(2.09)	1.15	(2.20)	–11.92	(3.07)	–3.88	329.0	+
	NY6-R2	0.94	(0.47)	16.79	(3.56)	–15.85	(3.87)	–4.09	353.5	+
Capitella Individuals	NY6	764.00	(250.98)	7.35	(2.10)	756.65	(231.85)	3.26	297.0	+
	R2-NY11	–33.81	(26.46)	–7.78	(1.02)	–26.04	(24.46)	–1.06	215.5	
	NY6-NY11	729.89	(243.98)	–0.54	(2.53)	730.43	(225.39)	3.24	293.0	+
	NY6-R2	763.70	(250.94)	7.24	(2.10)	756.47	(231.82)	3.26	298.5	+
Total Biomass mg	NY6	6,342.96	(1,117.70)	2,835.95	(269.15)	3,507.01	(1,072.76)	3.27	305.0	+
	R2-NY11	17,446.32	(5,082.75)	22,488.39	(2,839.86)	–5,042.07	(5,611.53)	–0.90	217.0	
	NY6-NY11	–9,685.54	(3,069.05)	–5,804.69	(761.82)	–3,880.85	(2,952.42)	–1.31	201.0	
	NY6-R2	–27,131.85	(3,374.24)	–28,293.08	(2,609.13)	1,161.22	(4,205.69)	0.28	211.0	
Adjusted Biomass mg	NY6	6,335.46	(1,117.13)	2,822.70	(268.52)	3,512.76	(1,072.07)	3.28	306.0	+
	R2-NY11	15,155.43	(1,013.95)	12,956.64	(2,065.50)	2,198.79	(2,423.59)	0.91	260.0	
	NY6-NY11	–324.31	(1,198.96)	–4,498.32	(352.62)	4,174.00	(1,171.42)	3.56	296.0	+
	NY6-R2	–15,479.74	(957.34)	–17,454.96	(1,996.28)	1,975.21	(2,334.37)	0.85	205.0	
Nemertean Biomass mg	NY6	2,876.09	(466.04)	922.79	(115.00)	1,953.30	(448.13)	4.36	332.0	+
	R2-NY11	34.37	(154.68)	257.73	(231.81)	–223.36	(288.70)	–0.77	244.0	
	NY6-NY11	2,636.93	(456.26)	695.73	(129.74)	1,941.20	(444.24)	4.37	336.0	+
	NY6-R2	2,602.56	(482.18)	438.00	(236.03)	2,164.56	(513.45)	4.22	326.0	+
P. affinis Biomass mg	NY6	184.48	(55.60)	224.16	(132.65)	–39.68	(152.47)	–0.26	231.0	
	R2-NY11	14,428.85	(839.25)	11,343.54	(1,654.15)	3,085.31	(1,950.79)	1.50	287.0	+
	NY6-NY11	–1,060.09	(238.47)	–1,065.41	(249.26)	5.32	(348.26)	0.02	192.0	
	NY6-R2	–15,488.94	(867.46)	–12,408.95	(1,632.47)	–3,079.90	(1,939.90)	–1.59	286.0	+
Tomato Seeds	NY6	984.56	(116.53)	1,233.10	(74.27)	–248.54	(134.34)	–1.85	251.0	
	R2-NY11	325.35	(58.70)	419.56	(66.64)	–92.21	(90.23)	–1.02	220.5	
	NY6-NY11	984.07	(116.46)	1,232.66	(74.24)	–248.58	(134.27)	–1.85	251.0	
	NY6-R2	656.72	(116.02)	813.10	(95.05)	–156.37	(148.55)	–1.05	231.0	

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[1] *Arctica* and *Pitar* excluded from total biomass.

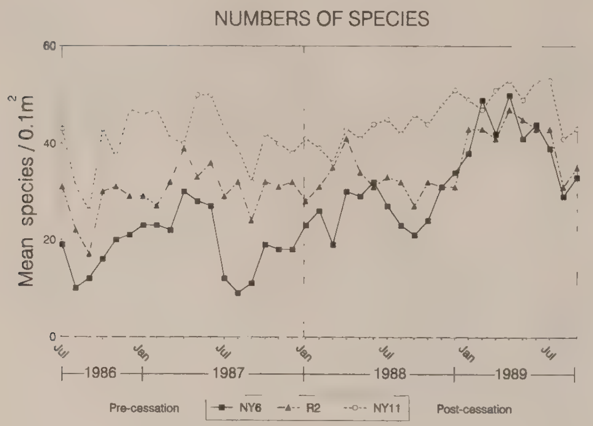

Figure 2
Mean numbers of species (all taxa combined) per 0.1m^2 at sampling stations in the inner New York Bight. Station NY6=square; station R2=triangle; station NY11=circle.

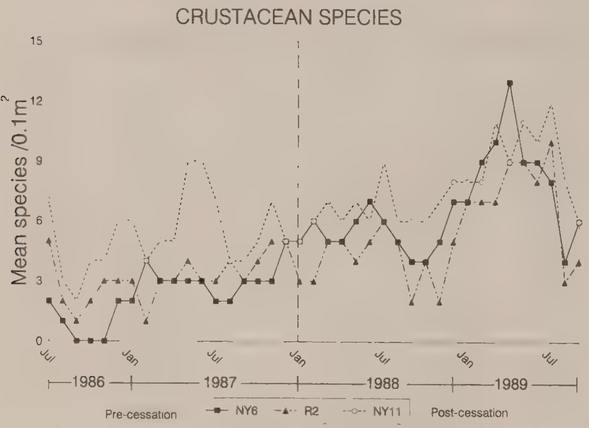

Figure 3
Mean numbers of species of crustaceans per 0.1 m^2 at sampling stations in the inner New York Bight at sampling stations in the inner New York Bight. Station NY6=square; station R2=triangle; station NY11=circle.

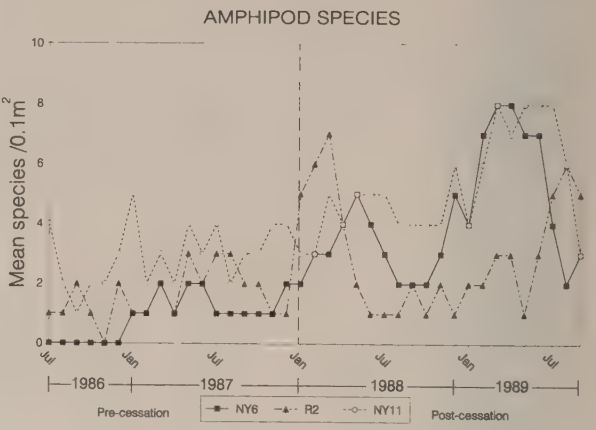

Figure 4
Mean numbers of species of amphipods per 0.1 m^2 at sampling stations in the inner New York Bight. Station NY6=square; station R2=triangle; station NY11=circle.

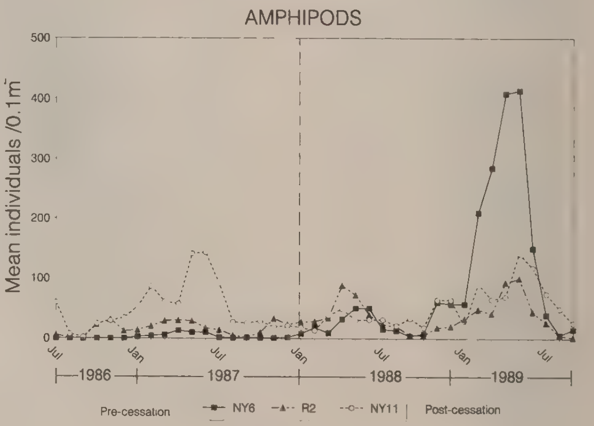

Figure 5
Mean abundances of all amphipods combined per 0.1 m^2 at sampling stations in the inner New York Bight. Station NY6=square; station R2=triangle; station NY11=circle.

amphipods (Fig. 4), and numbers of amphipod individuals (Fig. 5) relative to NY11 were confirmed for NY6 but not R2. The hypothesized decrease in abundance of *Capitella* spp. (Fig. 6) at NY6 was also confirmed. Table 1 also shows that, relative to NY11, there were significant increases in numbers of species of mollusks (Fig. 7) at both NY6 and R2, and increases in abundance of the amphipod, *Unciola irrorata* (Fig. 8) at NY6. There was a significant decrease in biomass of nemerteans (Fig. 9) at NY6 compared to both NY11 and R2. Overall biomass at NY6 decreased significantly (reduced biomass of nemerteans accounted for more than half the decrease). The decrease in adjusted biomass, with the bivalves *Arctica islandica* and *Pitar morrhuanus* excluded (Fig. 10), was significant relative to NY11, but the decrease in unadjusted biomass was not. Biomass of *Pherusa affinis* (Fig. 11) increased at NY6 compared to R2 but not NY11. There were no significant changes in numbers of tomato seeds (Fig. 12).

Cluster analysis (Fig. 13) indicated clear separation of the three stations, with all months at NY11 grouping at $Cz \geq 0.43$, all months at R2 at $Cz \geq 0.24$, and all months at NY6 at $Cz \geq -0.24$. NY11 and R2 joined at $Cz = -0.69$, and NY6 joined this group at $Cz = -0.97$. At the

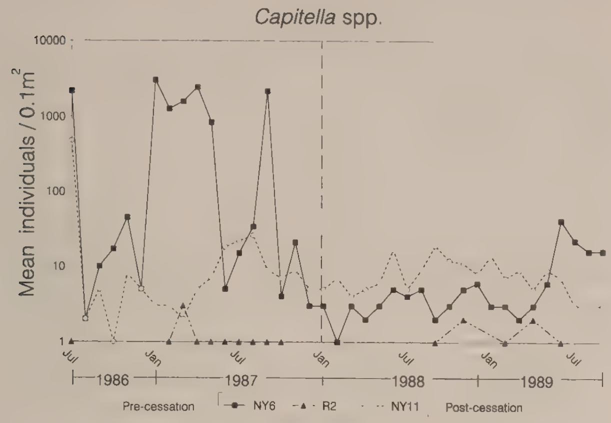

Figure 6
Mean abundances of *Capitella* spp. per 0.1 m^2 at sampling stations in the inner New York Bight. Station NY6=square; station R2=triangle; station NY11=circle.

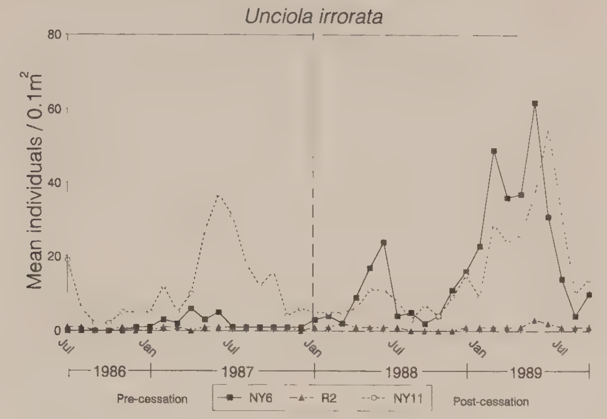

Figure 8
Mean abundances of *Unciola irrorata* per 0.1 m^2 at sampling stations in the inner New York Bight. Station NY6=square; station R2=triangle; station NY11=circle.

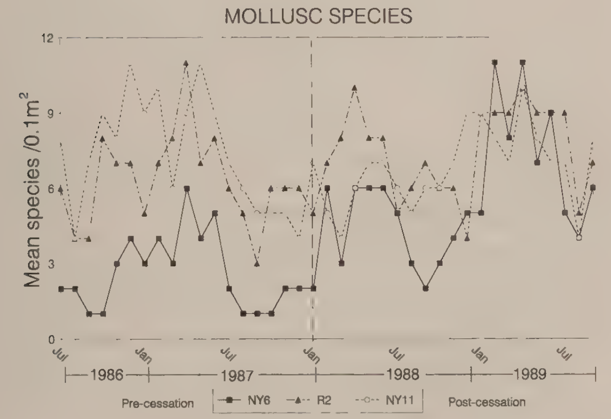

Figure 7
Mean numbers of species of mollusks per 0.1 m^2 at sampling stations in the inner New York Bight. Station NY6=square; station R2=triangle; station NY11=circle.

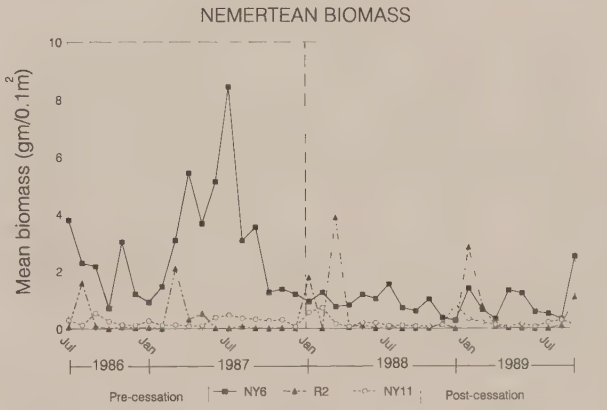

Figure 9
Mean biomasses of nemerteans per 0.1 m^2 at sampling stations in the inner New York Bight. Station NY6=square; station R2=triangle; station NY11=circle.

0.60 similarity level, NY11 and R2 each formed four groups of months, while there were nine groups at NY6. All postcessation months at NY6 grouped separately from all precessation months, and closer to R2. The other stations also segregated into pre- vs. postcessation groups with the exception of one NY11 group which contained samples from both July through December 1987 and January through April 1988, and one R2 group which included October through December 1987, January through December 1988, and July through September 1989.

Table 2 lists data from the October 1990 and March 1991 collections (i.e. 34 and 39 months after cessation of dumping) for the same variables examined in the above BACI analyses. These more recent data have not been compared statistically with those from the last 18 months of dumping and first 21 months after cessation, but some observations may be noteworthy. The postcessation trend toward increasing overall numbers of species at NY6 relative to NY11 and R2 appeared to continue, and in March 1991, NY6 had the most species of the three stations. Much of the increase was due to an influx of mollusk species. Postcessation trends toward reduction in density of *Capitella* spp. and biomass of nemerteans also continued at NY6. However, total, adjusted, and *Pherusa affinis* biomass remained consid-

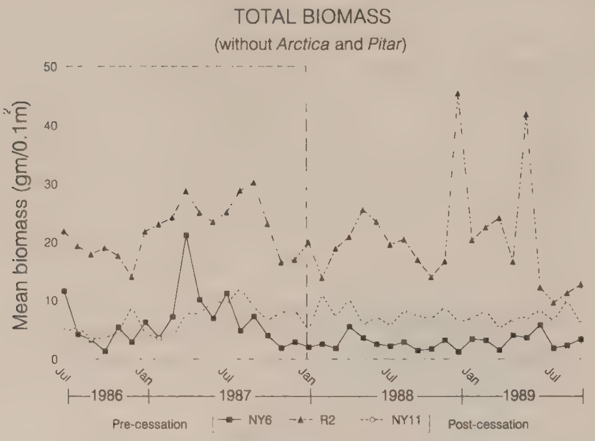

Figure 10
Mean biomasses of all taxa combined, excluding *Arctica islandica* and *Pitar morrhuanus*, per 0.1 m^2 at sampling stations in the inner New York Bight. Station NY6=square; station R2=triangle; station NY11=circle.

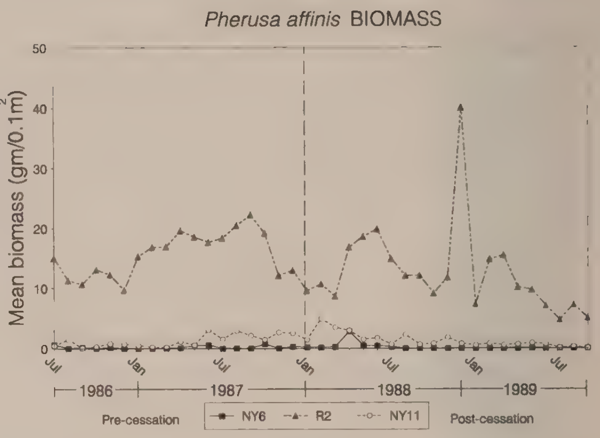

Figure 11
Mean biomasses of *Pherusa affinis* per 0.1 m^2 at sampling stations in the inner New York Bight. Station NY6=square; station R2=triangle; station NY11=circle.

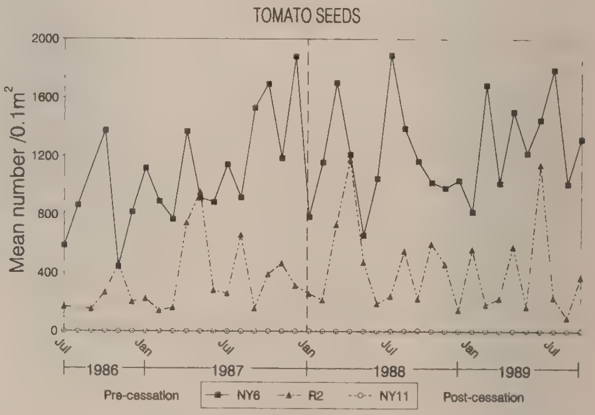

Figure 12
Mean numbers of tomato seeds per 0.1 m^2 at sampling stations in the inner New York Bight. Station NY6=square; station R2=triangle; station NY11=circle.

erably lower than values at NY11 and R2, while tomato seeds were still about as abundant at NY6 as in earlier surveys. At R2, overall (adjusted) biomass, *P. affinis* biomass, and numbers of seeds appeared to decrease.

Amounts of sludge detritus at NY6 were quite variable within and between the monthly cruises, but averaged an estimated liter per sample, with no obvious decrease in the first 21 months after cessation. Smaller amounts (about one half liter) were noted in the 1990 and 1991 collections.

Discussion

At NY6, the station chosen to represent greatest sludge effects, some variables showed clear responses within 21 months after cessation of sludge dumping, while other variables did not. The increase in numbers of species per sample at NY6 compared to both NY11 and R2 (Table 1, Fig. 2) is perhaps the clearest evidence of response. This variable is a useful indicator of environmental stress (Green, 1977; Chapman et al., 1987). The increases in numbers of species of mollusks, crustaceans, and amphipods at NY6 compared to the other stations (Table 1) can also be taken as indicators of habitat improvement, as these taxa are in general considered pollution-sensitive relative to the polychaetes which dominate much of the inner Bight (Pearce et al., 1981; Boesch, 1982). These variables, however, are subsets of overall species numbers, and so are somewhat interdependent and redundant as measures of response.

Changes in abundance of several taxa also signalled responses to cessation. The small deposit-feeding polychaetes *Capitella* spp. a complex of several species (Grassle and Grassle, 1976), have been used worldwide as an indicator of organic enrichment effects (Pearson and Rosenberg, 1978). *Capitella* spp. had been the overwhelming numerical dominants in most NY6 samples analysed in the 1970's and early 1980's (Reid et al., 1991). At the outset of the present study, many collections still had very high densities (over 1000 per 0.1 m^2) of these species, although densities of less than 50 per 0.1 m^2 were also common (Table 1, Fig. 6). Following cessation, mean densities never exceeded 50 per 0.1 m^2. Conversely, amphipods had been a very small component of the NY6 fauna during dumping, averaging

Table 2

Means of values per 0.1m^2 for three grab samples at each replicate station, October 1990 and March 1991.

	NY6		R2		NY11	
	10/90	3/91	10/90	3/91	10/90	3/91
Total Species	29.7	42.7	31.0	32.0	37.0	38.7
Mollusk Species	3.7	10.7	6.3	9.3	7.0	8.3
Crustacean Species	5.3	6.7	4.0	3.3	6.3	7.7
Amphipod Species	3.3	3.3	2.3	2.0	4.3	5.0
Amphipod Individuals	12.7	22.0	12.7	14.0	19.7	17.7
U. irrorata Individuals	6.0	2.7	0.3	0.3	3.7	6.7
Capitella Individuals	2.7	3.3	0.0	0.0	0.0	0.3
Total Biomass (mg)	3,324	3,282	7,792	55,954	7,995	5,515
Adjusted[1] Biomass (mg)	3,110	3,018	7,788	8,408	5,153	4,992
Nemertean Biomass (mg)	873	153	0.0	1.3	43.3	77.3
P. affinis Biomass (mg)	67.7	16.3	3,638	3,010	630	657
Tomato Seeds	934	1,221	176	139	0.7	0.0

[1] *Arctica* and *Pitar* excluded from total biomass.

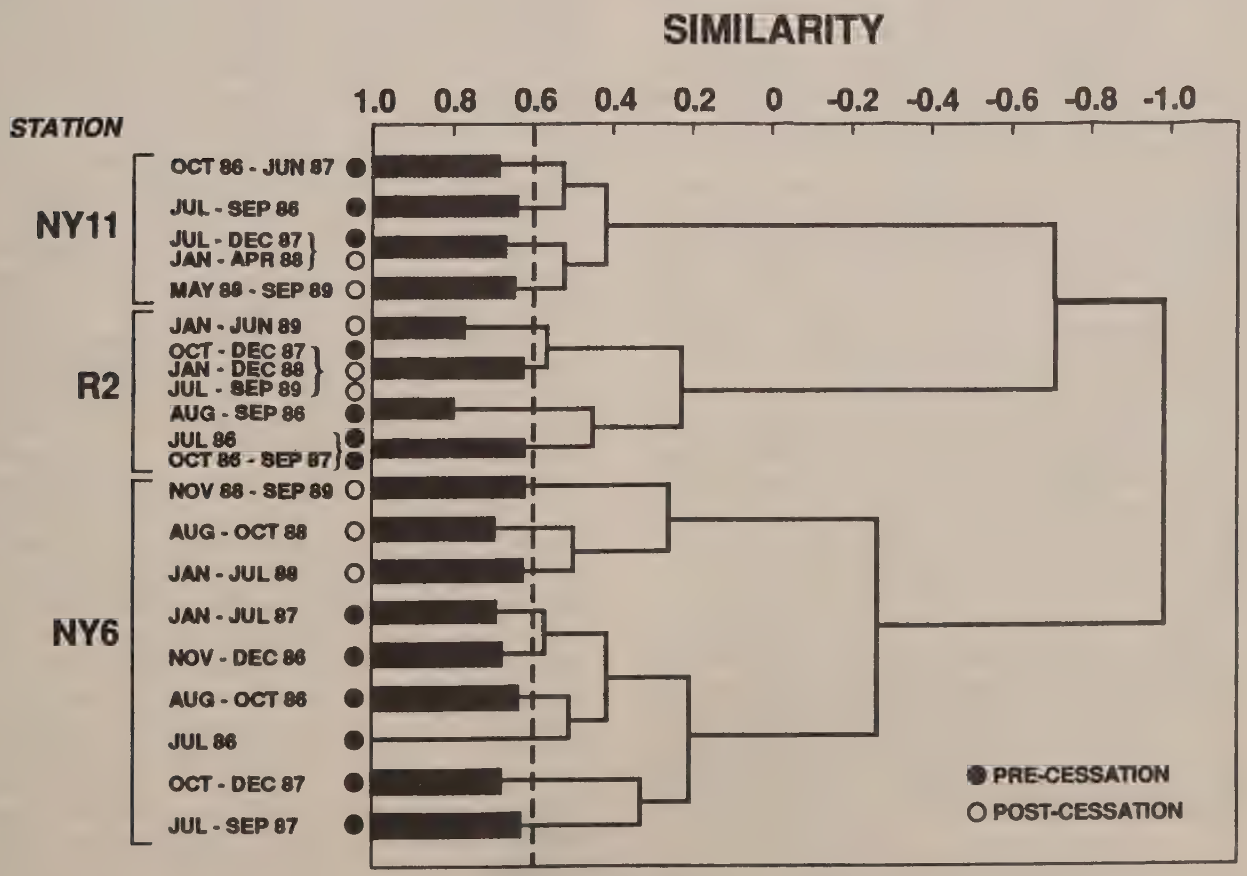

Figure 13

Dendrogram showing between-group similarities in species composition in the New York Bight, with groups formed at 0.60 similarity level. Each wider bar to left indicates the level of similarity of all months within a given group.

1.5 per 0.1 m^2 in 1980–82 and decreasing to 0.04 per 0.1 m^2 in 1983–85 (Reid et al., 1991). Most of the postcessation increase was due to *Photis pollex* (Fromm et al.[4]), and to a lesser extent *Unciola irrorata* (Table 1, Fig. 8); the latter had been the most abundant amphipod at NY6 during dumping.

Other measures did not show a rapid response of the NY6 fauna to cessation. If the environment at NY6 was indeed similar to R2 except for sludge influence, removal of that influence might be expected to lead to a total macrobenthic biomass similar to that of R2, which is representative of much of the Christiaensen Basin (Reid et al., 1991). There was little tendency for the NY6 biomass to approach the enriched level of R2, however (Table 1, Fig. 10). Biomass of the dominant species at R2, the polychaete *Pherusa affinis*, also did not increase noticeably at NY6 (Table 1, Fig. 11). *Pherusa affinis* was the most important prey for fish examined in the 12-Mile Dumpsite study (Steimle, 1995), so increased abundance of *P. affinis* at NY6 would have been an indicator that functional value of the area to predators was approaching that of the surrounding Christiaensen Basin. NY6 also showed little tendency toward increases in biomasses of the other dominants at R2 (e.g. the polychaete *Nephtys incisa*, bivalve *Pitar morrhuanus*, and anthozoan *Ceriantheopsis americanus*) or at NY11 (the bivalve *Nucula proxima*) (Fromm et al.[4]).

Cluster analysis also gave mixed indications of responses at NY6 (Fig. 13). That all postcessation months at NY6 grouped separately from all precessation months, and closer to R2, is taken as a response to phaseout. However, the postcessation months at NY6 were still most similar to precessation months there, and highly dissimilar from the faunas of R2 and NY11.

The continued high abundance of tomato seeds at NY6 (Table 1, Fig. 12) indicated that at least a refractory component of the sludge persisted following cessation of sludge disposal. An alternate explanation is that a deep reservoir of seeds and perhaps other sludge components was gradually being dispersed. However, sludge is not thought to have formed deep deposits in the inner Bight (Swift et al., 1976).

Seeds were still very numerous when NY6 was resampled 34 and 39 months after cessation (Table 2), and volumes of sludge detritus were only slightly smaller than during the 21 months immediately after cessation (authors' observations). According to some faunal variables, e.g. numbers of species and numbers of individual *Capitella* spp., NY6 might be considered to have responded completely to removal of sludge influence by 39 months after cessation. However, faunal biomass at NY6 remained sparse, with little increase in abundance of the NY11 and R2 dominants. These mixed indications of response are reasonable. All of the NY11 and R2 dominants apparently live more than two years, some perhaps more than ten years (Caracciolo and Steimle, 1983; Fallon, 1985; Steimle, 1985). Most have low mobility, so colonization by adults would be slow, and many species arriving as larvae may not have reached maximum biomass. If reductions in toxicants at NY6, e.g. sulfide (Draxler, 1995) and trace metals (Zdanowicz et al., 1995) have rendered the area suitable for colonization, several years may be needed to develop a mature "normal" macrobenthic assemblage. Even with reduction in contaminant concentrations, NY6 may be physically unsuitable for some species due to the remaining detritus.

At R2, numbers of tomato seeds appeared lower in the 34- and 39-month postcessation samples, in contrast to the persistence of the seeds at NY6. Overall (adjusted) and *Pherusa affinis* biomass also seemed substantially lower in the most recent samples. If this trend is real, one possible cause would be a reduction in the enrichment effect noted for R2 at the study's outset. The trend agrees with the reduction in sediment organic carbon reported for R2 after cessation (Draxler, 1995).

Other studies of effects of sewage disposal on benthic macrofauna give little indication of what rates of response to expect with abatement of disposal. At the Philadelphia dumpsite, sludge was dumped at lower rates and over a much shorter time than at the New York Bight site (Devine and Simpson, 1985). Accumulation of apparent sludge constituents and effects was not seen to the same extent as in the inner Bight. However, during dumping a slight increase in numbers of *Capitella* spp. and decrease in amphipod densities were reported (Lear and O'Malley, 1983). No postdumping macrofaunal data are available, but microbial sludge indicators returned to background levels in about four years (Reid et al., 1987) at this more dispersive site. Off Garroch Head in the Firth of Clyde, Scotland, a sludge dumpsite in use since 1904 had a depauperate, highly altered fauna when dumping ceased in 1974. The area was not sampled again until 1985, at which time the former dumpsite was similar to Clyde Sea stations remote from dumping in numbers of species and distributions of individuals among species (Moore and Rodger, 1991).

On the Palos Verdes shelf off southern California, the large quantities of primary-treated wastewaters discharged through outfalls had impacts on benthic fauna similar to sewage sludge effects apparent in the present study, e.g. in 1980 overall biomass and numbers of species were depressed 1 km from the outfalls, and *Capitella* spp. accounted for 85.6% of infaunal abundance (Stull et al., 1986a, 1986b; Swartz et al., 1986).

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[4] Fromm, S. A., D. Jeffress, J. J. Vitaliano, A. B. Frame, and R. N. Reid. In preparation. Responses of benthic macrofauna to phaseout of sewage sludge disposal in the inner New York Bight—Data report. U. S. Dep. Commer., NOAA Tech. Memo. NMFS F/NEC.

Improvements in effluent quality clearly reduced the severity of outfall impacts by 1983; biomass and numbers of species increased, and *Capitella* spp. dropped to 10.7% of total abundance. The extent to which faunal recovery was linked to the wastewater improvements could not be completely determined, however, as sediment reworking by an ephemeral population of echiurid worms also contributed to the observed changes (Stull, 1986a). From 1983 to 1986 there was further improvement in effluent quality, but the benthos exhibited complex changes. Biomass and numbers of species 1 km from the outfalls decreased, and abundance of *Capitella* spp. increased, to 1980 levels. Three km from the outfalls, benthic variables indicated basically no change between 1983 and 1986, while improvement was indicated 5–15 km from the outfalls. This complex pattern of change was considered due to interactive effects of wastewater and natural phenomena such as El Niño and winter storms (Ferraro et al., 1991).

One year after termination of discharge from the Hyperion sludge outfall in Santa Monica Bay, also off southern California, faunal responses showed some marked similarities to those seen in the present study. Those responses included a two order of magnitude decrease in numbers of *Capitella* spp. in the contaminated area and a failure of the dominant species in the reference area, the brittlestar *Amphiodia*, to return to the contaminated area (Southern California Coastal Water Research Project[2]).

The present study will continue periodically to sample the benthos of NY6, NY11, and R2 to follow longer-term responses to cessation of sewage sludge disposal. Relationships with other variables, such as sediment grain size, redox potential, sulfide and carbon concentrations, and fish diets, will be explored in an attempt to determine causes of the observed responses to cessation.

Acknowledgments

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Audience Questions

Question: The NY6 station is probably a depositional-silt station, and unless that turns into a sandy area like NY11, I do not think you are ever going to see a large population of *Nepthys* returning to that area. What I am driving at is that sediment type is probably one of the main determiners of what species composition will be.

R. Reid: I agree completely with that, and when we designed the experiment we tried to get similar sediment types. It looks like they are not all that similar. There are even surprising changes over time at a given station; we will sometimes find pretty sandy sediment at NY6 after a storm; the sludge or silt or whatever is there apparently is blown away or covered with sand temporarily. If NY6 is a more depositional area than the rest of the Christiaensen Basin, then maybe what I was suggesting in the beginning is true—the fauna will never look like the rest of the Basin.

Reduction in *Clostridium perfringens* and Fecal Coliform Bacteria in the Shellfish Closure Area of the New York Bight

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ABSTRACT

Parts of the inner New York Bight have been closed to commercial shellfishing since 1970 because of elevated levels of coliform bacteria from sewage sludge and other sources. As part of a study of responses to cessation of sewage sludge dumping in the Bight in December 1987, fecal coliform bacteria in surface and bottom waters were re-surveyed in October 1989. Sediment samples were also taken to determine concentrations of *Clostridium perfringens* spores, another microbial indicator of sewage contamination. Fecal coliform counts were below the detection limit (most probable number [MPN] of 9/100 ml) in all 31 surface water samples and 29 of 31 bottom water samples. The highest concentration measured was 139/100 ml in deep waters in the mid-Christiaensen Basin. Counts in general were much lower than when dumping was ongoing and lower than in many estuaries where shellfishing is permitted. Therefore, it may be possible to reopen much or all of the shellfish closure area, although microbiological as well as chemical contaminants in sediments and shellfish also must be considered before any decision is reached. However, the modest stocks of clams apparently present in the closure area would probably contribute little to the overall New York Bight harvests. Some of the observed reduction in coliform counts may be due to factors other than cessation of sludge dumping, e.g. year-round chlorination of sewage wastewater effluents. Concentrations of *C. perfringens* spores were lower than during dumping but were still markedly high over much of the Christiaensen Basin, in agreement with other findings from the study on responses to cessation of dumping.

Introduction

Numerous studies have documented elevated concentrations of fecal coliform bacteria (Verber, 1976; Gaines, unpublished data) and spores of *Clostridium perfringens* (Cabelli and Pedersen, 1982; Graikoski, 1982; Reid et al., 1987) in the apex, or northwest corner, of the New York Bight (Fig. 1). These bacteria are used as indicators of possible contamination by pathogenic bacteria and viruses. In 1970, the U.S. Food and Drug Administration (FDA) closed the area (Fig. 1) within a radius of 11 km (6 nmi) of the northwest corner of the 12-mile sewage sludge dumpsite to commercial shellfish harvesting after finding elevated levels of coliform bacteria in surface waters (Verber, 1976). In 1974, after increasing bacterial contamination was found in inshore waters, the closure area was extended to the Long Island and New Jersey shorelines (Fig. 1; Verber, 1976).

The shellfish closure was among the reasons cited for discontinuing the use of the 12-mile dumpsite; dumping was phased out between March 1986 and December 1987 (EPD [Environmental Processes Division], 1988).

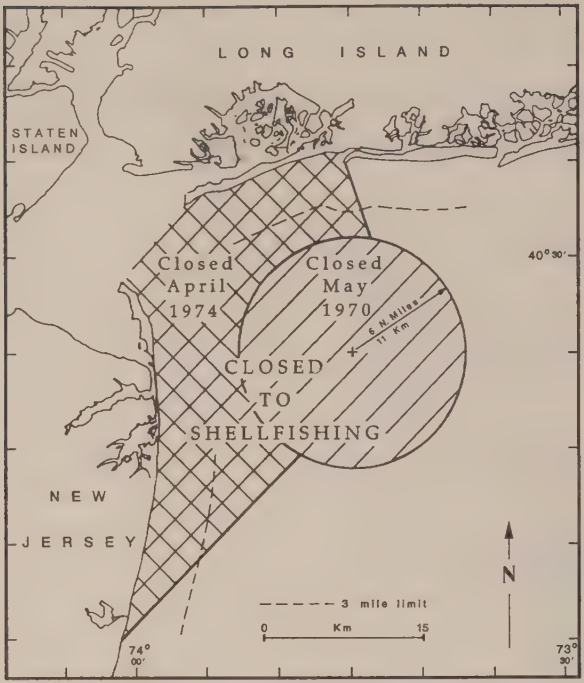
LONG ISLAND
STATEN ISLAND
Closed April 1974
Closed May 1970
6 N. Miles
11 Km
CLOSED TO SHELLFISHING
NEW JERSEY
40° 30'
N
3 mile limit
0 Km 15
74° 00'
73° 30'

Figure 1
Inner New York Bight, with 1970 shellfish closure area and 1974 extension. After Verber (1976).

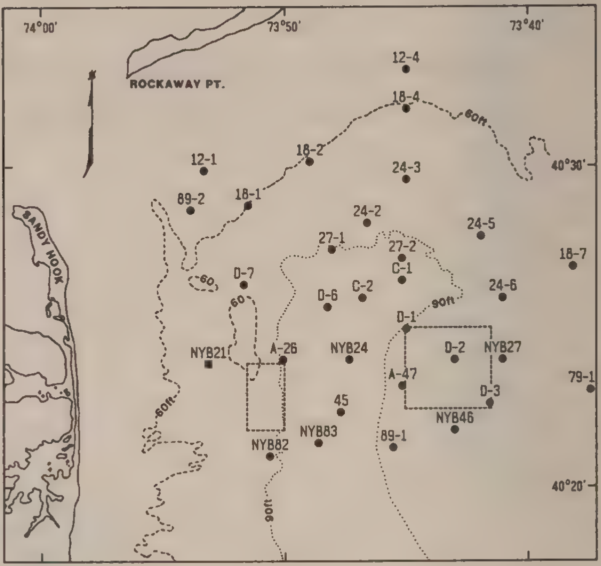
74°00'
73°50'
73°40'
ROCKAWAY PT.
SANDY HOOK
40°30'
40°20'
12-4
18-4
60ft
12-1
18-2
24-3
89-2
18-1
24-2
24-5
27-1
27-2
C-1
18-7
D-7
60
D-6
C-2
24-6
90ft
D-1
NYB21
A-26
NYB24
D-2
NYB27
A-47
79-1
D-3
45
NYB46
NYB83
NYB82
89-1
90ft

Figure 2
Station locations for fecal coliform and *Clostridium perfringens* sampling in inner New York Bight. Square=12-mile sewage sludge dumpsite, rectangle=dredged material dumpsite.

The dumpsite closure provided the impetus for a multidisciplinary study of responses to cessation of dumping (EPD, 1988). One hypothesis of the study was that contamination would decrease to the extent that shellfish beds could be reopened (EPD, 1988). This paper describes results of an October 1989 survey of microbial contamination in the closure area. Data are compared with those collected in similar surveys made while dumping was ongoing. Comparisons are also made with microbial data from other components of the study of responses to the cessation of sludge dumping (Davis et al., 1995; O'Reilly et al., 1995).

Methods

Samples were collected at 31 stations in the inner New York Bight (Table 1, Fig. 2) from 3–6 Oct. 1989. Station locations corresponded to those used in prior FDA and other microbiological surveys (e.g. Cabelli and Pedersen, 1982), with stations added in and near the sewage sludge dumpsite to provide a better description of concentrations of fecal coliforms and *C. perfringens* in that area. Surface water samples were taken with a 5-L Niskin sampler and bottom water samples from water overlying the sediment in a 0.1 m^2 Smith-McIntyre grab sampler. Sediment samples were collected by scraping about 200 g of surficial sediment from the grab sampler with a sterile tongue depressor. All samples were placed in sterile bottles. Water samples were analyzed for fecal coliforms aboard the vessel by using the American Public Health Association (APHA) A1M procedure and a 12-tube single dilution scheme (APHA, 1970). Each sample was inoculated in the test media immediately after sampling, incubated for 3 h at 35°C and then transferred to 44.5°C for an additional 21 h. Each tube was then read for the presence of gas or active effervescence in an inverted vial. Sediment samples were shipped to the FDA Shellfish Sanitation Branch's Northeast Technical Services Laboratory, Davisville, R. I., where they were analyzed for *C. perfringens* spores using iron milk media and most probable number (MPN) methodology (Abeyta et al., 1985).

Results

Table 1 lists all sampling locations, concentrations of fecal coliforms in surface and bottom waters, and concentrations of *C. perfringens* in sediments. Concentrations of fecal coliforms in surface waters were below the detection limit of 9.0 MPN/100 ml at all 31 stations. In bottom waters, fecal coliforms were detected at only two stations, C–1 (9.0 MPN/100 ml) and NYB24 (139

Table 1

Station locations (see Fig. 2) and bacteriological data expressed as Most Probable Number [MPN]) for inner New York Bight survey, October 3–6, 1989.

Station	Latitude	Longitude	Surface Fc[1]	Bottom Fc	Sediment Cp[2]
12–4	4033.10	7345.00	<9.0	<9.0	27
18–4	4031.80	7345.00	<9.0	<9.0	130
18–2	4030.20	7349.00	<9.0	<9.0	1,100
12–1	4029.90	7353.30	<9.0	<9.0	79
24–3	4029.60	7345.00	<9.0	<9.0	9,200
89–2	4028.60	7353.80	<9.0	<9.0	110
18–1	4028.80	7351.40	<9.0	<9.0	70
24–2	4028.30	7346.60	<9.0	<9.0	5,400
27–2	4027.10	7345.10	<9.0	<9.0	9,200
24–5	4027.90	7342.00	<9.0	<9.0	170
27–1	4027.40	7348.10	<9.0	<9.0	3,500
D–7	4026.40	7351.60	<9.0	<9.0	5400
C–1	4026.50	7345.10	<9.0	9.0	16,000
18–7	4026.90	7338.20	<9.0	<9.0	13
D–6	4025.60	7348.20	<9.0	<9.0	no sample
C–2	4025.90	7346.80	<9.0	<9.0	16,000
24–6	4025.90	7341.10	<9.0	<9.0	230
D–1	4025.00	7345.00	<9.0	<9.0	490
NYB21	4023.80	7353.00	<9.0	<9.0	16,000
A–26	4024.00	7350.00	<9.0	<9.0	16,000
NYB24	4024.00	7347.40	<9.0	139.0	>16,000
D–2	4024.00	7343.10	<9.0	<9.0	70
NYB27	4024.00	7341.00	<9.0	<9.0	790
A–47	4023.20	7345.20	<9.0	<9.0	1,300
45	4022.30	7347.70	<9.0	<9.0	16,000
D–3	4022.60	7341.60	<9.0	<9.0	180
79–1	4023.10	7337.10	<9.0	<9.0	33
NYB82	4021.00	7350.60	<9.0	<9.0	490
NYB83	4021.40	7348.60	<9.0	<9.0	3,500
89–1	4021.20	7345.60	<9.0	<9.0	49
NYB46	4021.80	7343.00	<9.0	<9.0	280

[1] Fc = fecal coliform bacteria (9.0 MPN/100 ml was detection limit of 12 tube, single dilution, A1M fecal coliform analysis).

[2] Cp = *Clostridium perfringens* spores per g wet wt.

MPN/100 ml) (Table 1, Fig. 3). Station NYB24 is in relatively deep water (37 m) just 3.6 km west of the sludge dumpsite and 2.7 km SW of the "highly altered" station NY6, which was the focus for much of the sampling to determine responses to cessation of dumping (EPD, 1988). Concentrations (MPN) of *C. perfringens* ranged from 13 to $>1.6 \times 10^4$/g sediment (wet wt), with the highest value also recorded at NYB24 (Table 1). Contours of *C. perfringens* concentrations (Fig. 4) indicate that high values ($>1 \times 10^4$/g) characterized much of the Christiaensen Basin as well as the dredged material dumpsite and the area to the west of that dumpsite.

Discussion

All fecal coliform concentrations measured in both surface and bottom water samples in October 1989 were at least four orders of magnitude below the highest values reported during dumping. The 1966–67 surveys which led to the 1970 shellfish closure found fecal coliform MPN's in excess of 2.4×10^3/100 ml in surface waters near the sludge dumpsite when no sludge tanker was discharging in the area, and 2.4×10^6/100 ml after sludge discharge (Verber, 1976).

Existing standards for the harvesting of shellfish for direct consumption are 70 total and 14 fecal coliforms per 100 ml in waters overlying shellfish beds, and 700 total and 88 fecal coliforms if the harvest is depurated before consumption (U.S. Department of Health and Human Services, 1990). In the October 1989 survey, concentrations of fecal coliforms over most of the survey area were well below these standards, and also below concentrations found in many estuaries where harvesting is permitted (e.g. Madden et al., 1986). The coliform concentrations indicate that much or all of

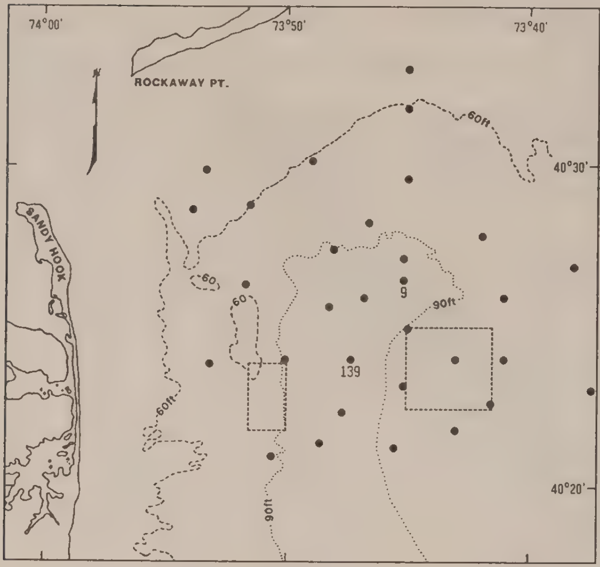

Figure 3
Sampling locations (dots) and densities (Most Probable Number [MPN]/100 ml) of fecal coliforms in bottom waters during 3–6 Oct. 1989. Where no value is shown, density was below detection limit (9.0/100 ml).

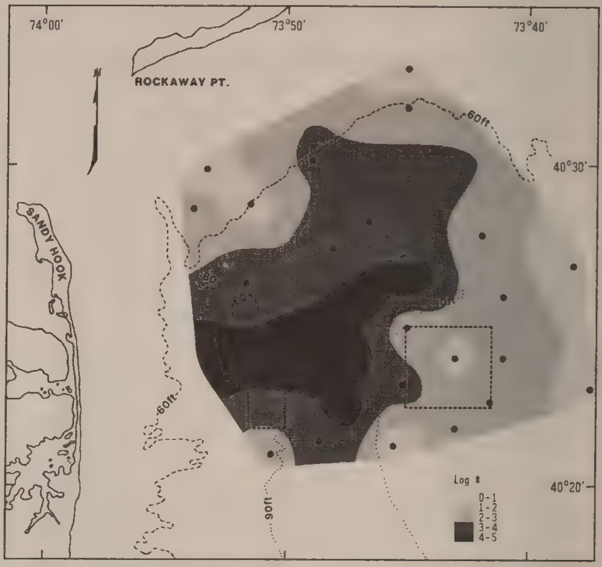

Figure 4
Sampling locations (dots) and contours of *Clostridium perfringens* spore densities in sediments ($\log_{10}$ MPN/g) during 3–6 Oct. 1989.

the closure area could be reopened. The single survey is considered adequate to characterize coliform concentrations because historical sampling has shown little seasonal change in concentrations (Gaines, unpublished). Areas with elevated coliform concentrations (e.g. around station NYB24) might remain closed, as might the dredged material disposal area. If any changes were made, revised charts would be issued indicating the new closure areas. The inner New York Bight is considered a unique situation, however, and FDA must evaluate both microbial and chemical contamination in sediments and shellfish before determining whether to change or remove the harvesting ban.

Trends in microbial contamination observed at other sewage sludge dumpsites after closure are of limited value in predicting recovery of the inner Bight. At an inshore site off Delaware, dumping ceased in May 1973 (Verber, 1976). The area was reopened to shellfishing 20 months later (January 1975) after monitoring indicated there was no longer significant bacterial contamination. The volume of sludge dumped at that site was only about 10% of that dumped at the inner Bight site, however (Verber, 1976). The sludge was subsequently dumped at the offshore "Philadelphia site" until November 1980. By June 1983 (31 months after cessation), that site met the criteria for shellfish harvesting and was reopened, based on concentrations of sewage-associated bacteria in sediments rather than in water (Reid et al., 1987). Again, the lower rate of sludge dumping, and the more dispersive nature of the Philadelphia site, make it difficult to extrapolate from this recovery period to the situation in the inner Bight.

If clams in the inner Bight have been chemically contaminated, depuration may be a long-term process. Tanacredi and Cardenas (1991) found no decreasing trend in polynuclear aromatic hydrocarbons (PAH's) in the hard clams *Mercenaria mercenaria*, which were contaminated experimentally and then held in clean water for 45 days.

Reopening the closed inner Bight beds would probably not lead to major increases in clam harvests. The few National Marine Fisheries Service (NMFS) surveys in the closure area have indicated relatively low abundances of Atlantic surf clams, *Spisula solidissima*, and ocean quahogs, *Arctica islandica*, although sampling has never been intensive enough to quantify abundances there (Murawski[1]). A 1985 surf clam survey in the "Sea Bright Borrow Area," which overlaps the southern inshore part of the closure area, disclosed very small stocks which would probably not support commercial clamming (U.S. Fish and Wildlife Service, 1986). That survey did collect substantial volumes of old shells, which may indicate historically productive shellfish beds with the potential to be productive in the future. Even if clams remain sparse, reopening the closure area would provide a tangible measure of improvement in the Bight's water quality.

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[1] S. Murawski, National Marine Fisheries Service, Woods Hole Laboratory, Woods Hole, MA 02543, pers. comm., Nov. 1991.

A possible drawback to reopening the area is loss of a "spawner sanctuary" of protected clams contributing to recruitment of future stocks. Such sanctuaries elsewhere (off Atlantic City, N.J., and Chincoteague, Va.), however, have not demonstrably enhanced clam recruitment. Enough spawning clams would probably remain in other closed areas (e.g. around outfalls) and elsewhere in the Bight that loss of the "spawner sanctuary" in the closure area would probably have a negligible effect on the supply of larvae. Larval supply in turn may bear little relation to recruitment, the determinants of which are poorly understood for both surf clams and ocean quahogs.

The observed reduction in fecal coliform counts over the entire survey area cannot be attributed exclusively to the cessation of sludge dumping. The coliform loading to the inner Bight from sludge was small compared to that from the Hudson–Raritan plume. It has been estimated that in 1973, sludge dumping supplied <0.01% of the fecal coliforms entering the Bight (New York City Department of Environmental Protection[2]). Some of the reduction in coliforms observed in the present survey may be a result of the extension of the period of chlorination of municipal wastewaters in the estuary, from summer to year-round, beginning in 1986. The Interstate Sanitation Commission[3] considered year-round chlorination to be the main factor in allowing a three month extension of the seasonal certification of surf clam beds off the Rockaways for harvesting for human consumption in 1987; in December 1988 the area became certified year-round (Interstate Sanitation Commission[3]). In the immediate vicinity of the 12-mile dumpsite, however, the extremely elevated coliform counts observed during dumping must have been largely due to sludge inputs, and cessation of dumping was undoubtedly responsible for most of the subsequent decline in counts (O'Reilly et al., 1995).

In contrast to fecal coliforms, spores of the sewage indicator bacterium, *Clostridium perfringens*, were still abundant in parts of the inner Bight 21 months after cessation of sludge dumping (Fig. 4). This agrees with the data of Davis et al. (1995) and O'Reilly et al. (1995); the latter authors predict it might take 15–20 years after cessation of dumping for spore densities in surface sediments to reach background levels. The persistence of *C. perfringens* is also congruent with the continued high abundance of tomato seeds and other sludge detritus reported by Reid et al. (1995), who note that the slow response of these variables could be a result of the ineffectiveness of erosive forces in winnowing large particles from surface sediments, and/or continued abundance in subsurface sediments as they are exposed by erosion.

The spatial pattern of spore abundance, with higher concentrations in the relatively deep, fine-sediment Christiaensen Basin, matches the pattern observed for other sewage indicators measured during the multidisciplinary study, e.g. sediment trace metals (Zdanowicz et al., 1995), carbon, sulfide, and redox potential (Draxler, 1995). Our data augment those from the standard sampling pattern for *C. perfringens* (O'Reilly et al., 1995) by extending coverage further to the north and west, and the October 1989 data help fill the gap between the April 1989 and March 1991 surveys. Results of our study differ somewhat from the O'Reilly et al. (1995) data in documenting some of the highest spore densities in and west of the dredged material dumpsite. Data from the two studies cannot be strictly compared, however, because of differences in methodologies.

Spore concentrations in the inner Bight, although elevated, were considerably lower in the 1989 survey than when dumping was ongoing. In a 1980 survey, the highest count recorded was 1.1×10^6 spores/g sediment (wet wt) at the northern edge of the sludge dumpsite (Graikoski, 1982). Broader-scale surveys of the Northeast shelf from 1978–82 reported highest spore counts for the entire region, $>1.0 \times 10^5$/g, just west of the sludge dumpsite (Reid et al., 1987). Thus, *C. perfringens* spore counts decreased roughly one to two orders of magnitude in in the inner Bight between the latter years of sewage sludge dumping and the FDA survey 22 months after cessation. Comparison with the decrease of four to six orders of magnitude in fecal coliform counts over the same period illustrates the much more conservative nature of *C. perfringens* as a sludge tracer.

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Audience Questions

Question: Is there anyone here willing to say what we do know about clam tissue residues?

R. Reid: Of pathogens, or pathogen indicators, or toxics?

Question: Of either pathogens or any indicator organisms of microbiota. Are we anywhere with respect to knowing of any changes in those reservoirs?

R. Reid: I can not answer that. The New York Bight Restoration Program has a pathogens working group that is dealing with this now. You may know that fecal coliforms are an indicator of pathogens. They may be a very poor indicator, and they may behave a lot differently than the pathogens themselves, so that is why the Restoration Program thought they should look at that. I really do not know what data are available now.

Future Research Directions in Ocean Disposal of Waste in the New York Bight—Fisheries Implications

Participants

Robert Tucker, Chair, Director, Division of Science and Research, New Jersey Department of Environmental Protection and Energy, Trenton, NJ;

James Chambers, NOAA, Silver Spring, MD;

William Gordon, Director, New Jersey Sea Grant College Program, Sandy Hook, NJ;

Frederick Grassle, Director, Institute of Marine and Coastal Sciences, Rutgers University, New Brunswick, NJ;

Joel O'Connor, U.S. Environmental Protection Agency, Region II, New York, NY;

John Pearce, Acting Director, Science and Research, Northeast Fisheries Science Center, Woods Hole, MA;

Cindy Zipf, Director, Clean Ocean Action, Sandy Hook, NJ. Introduction and closing remarks by R. A. Murchelano, Chief, Environmental Processes Division, Northeast Fisheries Science Center, Woods Hole, MA.

Introduction

R. A. Murchelano: Over the past day and a half you have heard intensive conclusions and summaries presented in the physical oceanography, chemical processes, and biota sections. As I was listening to the presenters, I made some notes that I would like to share with you.

No one in this room probably agrees totally with everything that has been said. I do not, and I am certainly not knowledgeable, by any means, with all the disciplines. Yet, I am also sure no one in this room disagrees that a relevant number of scientifically valid observations have been made pursuant to the cessation of dumping in relation to the environment and fisheries of the New York Bight. I doubt any of you disagree with that statement. Only members of the Environmental Processes Division and I, however, know the constraints under which this study was conducted.

I am incredibly proud of the accomplishments of this group. I feel that this Symposium is another milestone in the completion of the study, which will be followed by publication of a peer-reviewed volume that will provide a useful source of information for environmental and resource managers relevant to this topic for quite a period of time.

This study is an example of what can be done by identifying a discrete unit of research, which has been well planned, is interdisciplinary, and has a finite course. This pragmatic aspect of research, which is finite, which is time dependent, and which is well-defined, is something that I favor.

There is no way that I can express my thanks to all of you, specifically Mert Ingham, Anne Studholme, and Jay O'Reilly—the individuals that I have dealt with since 1986 who provided the identification for this study. When the study is completed, the publication of the symposium volume will attest to the utility of the study and to the effort that has been put into it.

Today's panel Chair is Dr. Robert Tucker, Director of the Division of Science and Research for New Jersey's Department of Environmental Protection. He will lead the panel discussion on the future implications for fisheries of ocean dumping.

Opening Remarks

R. Tucker: It is a real privilege for me to be here and to have this very distinguished panel present. I would like to add my congratulations to the staff at Sandy Hook and to their collaborators for this study. I came to Sandy Hook in 1971 and worked with Jack Pearce and some of my colleagues, who are still at Sandy Hook, on a survey of the dumpsite, funded by the Corps of Engineers. Back in those days I think we had more resources to devote to the kinds of studies than we have now. And, as has been indicated, the group at Sandy Hook and their co-workers have, with limited resources but with a great deal of effort, kept after these questions. I think it is a tribute to their work to have the presentations that we have heard. We have seen a lot of questions raised, and I am hoping that from the reactions of both the panel and the audience, we can address some of these.

I have asked the panel very briefly to react to what they have heard, and then we will try to be as interactive as we can. I would like to go directly to Jack Pearce who also has dealt with questions of this kind for many years.

J. Pearce: First of all, I too, have a good deal of praise for the people who conceived this study based on testable hypotheses which may enable us to put a lot of nails in the casket of ocean dumping, at least ocean dumping of sewage sludge.

As I spoke with people in Woods Hole or in Washington, a lot of academics, government people, managers, and others increasingly look to ocean dumping as a way of getting rid of some of mankind's more troublesome waste. Perhaps sewage sludge will be seen in the near future as a commodity too valuable to dump in the oceans, or to put in Texas landfills, or to incinerate on the shores of Newark Bay. The fact is, many people are concerned about what we are going to be doing for agricultural nutrients in the coming decades of the 21st century. For that reason alone I think it would be foolhardy to think about putting sludge in the sea.

But there are people who are looking at disposal of low level radioactive wastes, hot radioactive wastes, and a range of toxic substances that come from the industries that produce much of what you see in this room. All of the plastics, carpeting and so on, involve manufacturing processes producing a wide range of waste materials, some of which are not easy to get rid of even if you have recycling. Many people think that the ocean may be a proper receptacle for waste.

This study that we have heard in toto is better than anything I have seen in three or four decades in terms of what it tends to indicate and the objective nature of the study. It may set a precedent and be used as a benchmark. I feel that where we used to focus on problems such as ocean dumping or combined sewer outfalls, increasingly it turns out to be something called development or "the way we are." Not the way we were, but the way we are and the way we will be in coming decades.

As a society worldwide, Americans, Europeans, and Orientals are inclined to want to have an ever higher standard of living, but when you get together as a group of scientists and say what are the solutions to pollution, invariably it comes down to two things. One is population. Something will have to be done in the near future about ways to control population, or we are not going to have any solutions to our major environmental issues.

The second has to do with the way we live, and that is tied to development. Some demographers and economists are saying that, for a variety of reasons, many people are moving to the coastlines, be it the Great Lakes, the Gulf of Mexico, the Pacific, or the Atlantic. There are good indications that we may be headed for another series of dustbowls—not just in terms of global climate change, but just natural warming trends. If that is the case, it is predicted that more and more people will move to fresh and salt water environments, principally to have access to water. One of the sheiks in Saudi Arabia said that oil is the big issue today. It is the thing we fight for, but he said that in a few years what we will be fighting for is water. That should tell us all something—that we will be moving to those areas that have water and there will be ever greater pressures to develop the coastal zones.

That is what we have to be aware of now, and this study, along with future studies, should continue to involve collaboration with EPA, NOAA, the Food and Drug Administration, and the Fish and Wildlife Service, plus State agencies and academia. In coming decades we are going to have studies of this kind that will have a generic property to them. They will allow us to interpret the consequences of man's activities and provide the basis for true environmental management. We are not going to stop development anywhere in the world, least of all in the coastal zone. But we must have an understanding of what ocean dumping or non-point source pollution might do as it is carried forward in ever greater quantities or amounts. However you want to measure it, we will be looking to this kind of cooperative study perhaps as a classic example of how to do things well.

Finally, to do all of this we are coming down to a matter of public education. I have been chairing a group called Water Quality 2000—the challenge group on solutions to pollution. We have about 30 people, all eminent scientists from the Midwest, the East Coast, and the West Coast. The one thing they have in common is some experience in dealing with pollution in aquatic systems. In the end, when examining possible solutions, they recommended controlling population, controlling life styles, and using education as the tool to change people's attitudes.

Politicians and administrators are people that are responsible for agencies and government entities. By and large they are loathe to take a new step without public support. It will be our duty to increase knowledge through objective studies such as what you have heard today. Not only is the Water Pollution Control Federation, which sponsors Water Quality 2000, taking that view, the United Nations Environmental Program, a group that I have worked with over the last few years, feels that education is the principal way forward.

Bill Gordon is involved with a group called Fishermen Involved in Saving Habitats, a good example of an interest group or power cluster, whatever you want to call them, a group that has something that is worth a great deal. Such groups have to work to educate other people in the world.

Finally, we have just heard about how this study was done on a shoestring. I am here to tell you that these studies and similar studies will continue to be done on a shoestring. The social issues facing this country are so great today, that whether you are looking at AIDS, the homeless, education, or environmental studies, particularly in small agencies such as NOAA, that it is going to be harder and harder to find ever greater amounts of money. We are going to have to do more and more with less and less.

R. Tucker: Thank you very much, Jack. You have raised some really interesting questions. Bill Gordon has had years of experience with fisheries. Bill, can you give us your perceptions of where we are and what we might need to do in the future?

W. Gordon: Surely. I will simply echo Jack's thanks to those who put this conference together. It certainly is outstanding and well done. I want to come at it from a somewhat different perspective, perhaps, and throw out some statistics that might be useful to you. The present United Nations' estimate of global population growth is that it will increase from about six billion at the turn of the century and perhaps stabilize at ten billion in the middle of the next century. Before this happens it is apparent that the terrestrial environment is going to be excessively stressed.

If even 75% of the [population] that Jack is predicting will eventually reside within a coastal zone, the impact will be excessive. Moreover, currently and well into the future, 100% or so of our water supply is going to come from groundwater and other sources. Roughly 90% of our protein that is consumed comes from the land. I do not think we will be into aquaculture or desalination to provide the massive quantities of water and food necessary. The land then must be protected. Even if we turn to aquaculture, we still need high water quality on the edge of the sea or in freshwater if we are to use those products as a source of human food.

The challenge is clearly that of reducing waste, managing the residuals, and understanding the impact of those residuals that are left after all this high technology on the air, land, and water resources. We must define the role that oceans should play in the global waste management strategy. It is foolish to believe that any amount of technology or technical progress will eliminate the requirements for waste management disposal.

If my prediction of the future is correct, the marine scientists will be called upon to assist in the selection of disposal sites. This time I would hope that we, as scientists, will be prepared to respond with good solid facts and to actively participate in that decision as scientists, earning the respect of our citizens as we do that.

Clearly more research will be needed, and it should include a full range of physical and chemical oceanography. You will want to know the sites of relative stability. You will want to know the sites of high activity, erosion, and deposition and their biological aspects. We have heard many times at the conference these last two days that we do not know what took place, but here is our estimate of what the background noise is. It would have been far better if we had had excellent baseline studies of what was there in 1900 before we started dumping rather than an estimate of the background. The abundance and distribution of fishes and other animals, certainly the effects of fishing, could mask any knowledge of what the recovery rates may be in these areas. We simply do not know, because we do not understand the effects of fishing upon species richness. If we do not know what was there before, we do not know what the changes have been as a result of dumping and other recovery rates. We must understand the diets and critical habitat needs of these animals and, obviously, the implications of parasites and disease, not only in the animals themselves, but also upon human health.

I certainly strongly endorse public education; not only for our youth which are our future, but for our political leaders of today and the future. Good education can convert powerful public will to find those resources, to motivate political action. If we are not prepared as scientists to put and devote some of our efforts towards public education, then we have failed as scientists.

R. Tucker: Thank you very much, Bill.

Cindy Zipf is Director of Clean Ocean Action. Both Jack and Bill mentioned the importance of education. I can think of no one who has done more to educate the public in New Jersey and the surrounding area about the need to take care of the ocean.

C. Zipf: Thank you, Bob. I also want to commend the Sandy Hook group and the people that presented papers over the past two days. I think the exciting work that has been done shows much encouragement for those of us who are ocean advocates. I do have a little bit of history out at Sandy Hook. In fact, I began as a volunteer in the Behavior Department out at the Sandy Hook Marine Lab, so they can be blamed for whatever I am today.

With the wonderful news at the conference that there seems to be an ocean rebound out in the dumpsite area, I am struck by what I recall of my history in the last six years or so that I have been working on these issues.

I want to list some of the statements that were made in the past by scientists, regulators, and policymakers. One was "sludge dumping does not affect the ocean at the 12-Mile site." Another one was "sludge at the 106-Mile site will never reach the bottom." Another one that I recall: "Is the 106-Mile site area bottom a desert?" And then of course, we all remember the infamous "dead sea" 12 miles offshore. I think that what those statements and what science we have heard about today indicate is that we really do not know a lot about the ocean. We need to expand on the existing science—we simply do not know everything there is.

From a citizen's standpoint as well as an ocean advocacy standpoint, we do not view the ocean as a disposal medium. Putting it "out of sight and out of mind," or "the solution to pollution is dilution" is not the answer. I think the New Jersey experience attests to that.

In 1984, when Clean Ocean Action first started there were eight ocean dumpsites. Today there is only one without an end in sight and that is the mud dump. I think that the evidence from the last two days suggests that we should be looking at the mud dumpsite as well.

The New Jersey experience also shows that improvement for ocean water quality means improvement for the overall environmental arena. For instance, the ending of sewage sludge dumping in New Jersey has put added pressures on prevention of contaminated sewage sludge. If we are going to implement alternatives for sewage sludge, if we are going to use it as a fertilizer, it needs to be clean.

This emphasis on cleaning up sewage sludge will reduce industrial pollutants, require industrial users to implement preventative measures, source reduction, recycling, and other methods, so that the end of ocean dumping of sewage sludge is an improvement for overall waste management. Clearly the emphasis has to be on prevention.

Some of the other research items that I think need to be explored are expansions on the numbers that we have. It would be a shame not to expand on the studies to look at recuperative abilities, to see whether or not there is an expansion of the number of species in the benthic community at the Christiaensen Basin. We should see whether there is some additional improvement over time, and look at the body burden levels of the toxic pollutants, not only from the sewage sludge standpoint, but also from the dredge material dumpsite standpoint.

There were interesting facts in the physical oceanography section, indicating that there is some effect of wind on the Christiaensen Basin at the bottom. What effect does that have on the mud dump that rises like a grotesque wart 45 feet above the ocean floor?

Some of the other concerns or other research areas should concentrate not only on baseline data, but also expand information on the beneficial use of ocean biota and ocean processes for humankind. Not necessarily to determine areas for dumping but areas of production—to help solve some of the social problems we have on land.

To end up, I would like to say that I very much respect science. At Clean Ocean Action we utilize science to motivate rulemakers, either regulators or politicians, to emphasize that we need to get to the sources of pollution problems, whether they are on land or at sea. To close that loop I look forward to opportunities like this to work with scientists. I have worked with many of the members of this distinguished panel and, citizens and scientists, we try to work together. The goal is to help motivate and make correct decisions; not knee-jerk decisions, not "let's put it out in the ocean and hope it never comes back." Together, we should make well-founded scientific decisions, and I believe that those decisions will get back to the point of prevention of pollution problems.

R. Tucker: Thank you, Cindy. You have raised some excellent points, particularly the pollution prevention point. The Assembly in New Jersey just passed a Pollution Prevention Bill Monday [17 June 1991] which really does emphasize source reduction of toxics. Someone who has also been involved in ocean problems, in the days of the MESA [Marine Ecosystems Analysis] program and now currently with EPA, is Joel O'Connor. I know he has been instrumental in pushing for a wider or more diverse approach in the Harbor Estuary Program to make sure that we include habitat concerns. It is a pleasure to have you on the panel, Joel.

J. O'Connor: I recall it was several years ago now that I got a grandiose, almost outlandish, strategy for this dumpsite effort from Bob Murchelano and was very impressed with it. Certainly I was doubtful that Bob could pull together the resources and stay the course as he said. I am certainly now much more impressed with the outcome than I was with the plan. It is typically the other way around with plans and outcomes that I review. I would encourage, primarily, that the group at Sandy Hook and your colleagues from outside NMFS stay the course of even further assessment than you demonstrated yesterday and today. As I understand it, there is quite a bit of data which need to be analyzed, interpreted beyond what we have seen in the last two days. A few selected measurements in the future would be most useful.

Perhaps one obvious observation that I am sure the investigators have thought of is to aim for further interpretation by assessing on broader space scales and over longer time scales than we have typically discussed here. I think observations further out on the shelf made a decade or so before sewage sludge dumping, added to the five or six years worth of measurements, are very helpful in interpretation.

One of the things that I am still a little confused about is the dynamics of the surficial sediments and how well those are interpreted at this stage. It seems to me that a clear understanding of those dynamics underlies an understanding of a lot of other things that have been discussed. Another important influence that deserves more assessment is that of the confounding impacts of the mud dump on the impacts of sewage sludge.

One of my suggestions for further assessment would be to deemphasize efforts to determine the sources of effects, inasmuch as I expect those are going to be opaque for decades in most cases. Emphasize, instead, the regional effects of sewage sludge and the mud dump and their rates of change over time.

You have already gone quite some way toward documenting the effects of sewage sludge dumping, and these will be valuable social indicators for use in the future. We have a much clearer assessment of the effects of this particular dumping activity over decades.

I emphasize the importance of informing the public and the decision makers of what you have learned. I think this is perhaps even more important in this situation than documenting in peer-review journals. Simple sorts of fact sheets or other approaches that educational groups can outline for you would be a particularly valuable outcome of the whole activity. If you could really educate people about sewage sludge effects in the New York area that would be, I think, a fantastic contribution. Finally, I think all of the participants at the Sandy Hook Lab and outside collaborators can take a great deal of pride in the outcome, and I look forward to seeing even further assessments as you carry this further.

R. Tucker: We are also very fortunate to have Jim Chambers here from NOAA, Washington. With your concerns about habitat protection I would like to hear your reactions to what has gone on, and maybe you have some words of wisdom for how we can help the Fisheries Service continue the kinds of research that we have heard about for the last couple of days.

J. Chambers: I would agree with virtually everything that is been said. I took a little different tact when I starting thinking about the opportunity to speak here today, and I based that on our stewardship obligation for living marine resources—commercial, recreational, and ecologically important species throughout their range.

In the Northeast, a very large proportion, 41%, of the commercial catch for which we have good data, are estuarine dependent. My suggestion and counsel would be to think alot broader in terms of our responsibility. I would offer the priorities that exist within NOAA and within the National Marine Fisheries Service—the highest priority being threats to living resources—as a guide to what Sandy Hook and the Northeast Center ought to be concentrating on in their habitat research.

The top four priorities, nationally, are physical alteration of estuarine and riverine systems, contaminant loadings to estuarine and riverine systems, nutrient over-enrichment to those same systems, and finally, diversions of freshwater away from nurturing estuarine systems. These are the same top four priorities for the Northeast Region and Center which have been established and reestablished twice in the last decade. If these become the blueprint for where we ought to be targeting effort, suggestions for what we ought to be doing would fall roughly into four categories. First, a quantification of how many acres of major habitat types we have, how fast we are losing them, and the causes of loss. Secondly, a determination, using our research capability, which was amply demonstrated here over the last two days, [of] the functional contribution of the major habitat types to living resource productivity. Thirdly, determining what man's activities are doing to those critical habitat types, whether it be alteration of flows or wetland destruction, degradation, contaminant loadings, but focusing on those factors that are limiting future population sizes. And finally, determining how well we can restore habitats that have been degraded, using funds which are becoming available to us from those that have polluted in the past and from whom we are recouping damages. All those funds are to be applied to resource restoration, habitat restoration, etc.

So, my advice would be to think a lot more broadly in terms of a mission for this particular laboratory (e.g. Sandy Hook), in this particular division, but in essence the whole Northeast Center towards those priority threats.

R. Tucker: We in New Jersey and this particular area have been particularly fortunate to have Fred Grassle come from Woods Hole to take charge of the new Marine Sciences Institute at Rutgers. Fred, what do you predict both in terms of the research going on here and the interactions with Rutgers for the future?

F. Grassle: I would like to add my congratulations to the investigators involved in the work presented. It really is an excellent piece of work. I think it is very important that the people involved had a lot of experience in this region. I think it shows what can be done on a relative shoestring in a particular place with a focus of appropriate hypotheses.

The issue of monitoring the marine environment is going to be an increasingly important and difficult problem. Joel has already alluded to it. The fact is that we do not have the common sense view of the marine environment that we have in terrestrial areas. You can walk through a terrestrial area and have a pretty good idea of what is going on, but we do not have that advantage in most marine environments. And the challenge, of course, is to have large spatial and long timescale coverage. The fact is that in the financial climate that we are in, that is with shoestring budgets, we do not have the luxury to know what is going on with the marine environment as a whole. So we have to focus on places where we have a chance of understanding basic processes and then try to infer what is happening in other environments.

It seems that this area in the New York Bight apex is particularly interesting in this regard. It is certainly one of the most heavily impacted parts of the continental shelf, as opposed to the North Sea or the Baltic. It is also a very interesting area from a point of view of sediment dynamics, one area that is important to follow. We clearly have pulses of resuspension; we have heavier sedimentation in some areas than in others but with no accurate measurements. There are relations to topography in ways that are not obvious. There is the possibility of getting really high resolution of topography in areas like this to make better predictions, to get better measurements of shear stress to predict resuspension, and to measure rates of burial in the system. I think all of these processes could be improved.

I was delighted that many of the biological results made more sense than some of the chemical results. Particularly in my own area of specialization, the macrofaunal benthos seem to make a very clear-cut picture. I am one of those that like cluster diagrams and it seemed particularly clear to me that the macrofaunal benthos is responding in a very sensible way to what is happening (see Reid et al., this volume). So often in monitoring programs, it makes no sense. I agree with Joel and Cindy that something needs to be continued in this area.

I think that it is possible to make some inferences over larger areas from single sites, but there is a challenge to say how many sites, how big an area the extreme is. I go to meetings now on biodiversity partly because of the high biodiversity of the deep sea. This involves talking with lots of terrestrial ecologists. I was at one two weeks ago at the World Bank, where they said what we ought to do is what we did for the Amazon. We are just going to put it all on a GIS (geographical information system) and set the priorities, one through ten, for the entire environment. They are surprised when you tell them that there really is not that sort of database for marine systems, that we are lucky if we know a lot about biodiversity in a handful of places.

It seems to me that a lot of progress has been made towards learning about that in this dumpsite environment. I think the studies of flounder were very interesting and that there is some significant work done on the basic fisheries biology in the area.

General Discussion

R. Tucker: Several people have raised the need for monitoring or looking at the questions over a wider area. Jim just said that we ought to be looking at other parts of the habitat and earlier Jack raised the question of whether we, in fact, caused a general degradation of the area such that it is harder to pick out some of the changes. We have obvious changes in the benthos and the sediment chemistry, but it is less easy to pick up changes in fish and some of the other migratory species.

J. O'Connor: Do you mean there may be indications of degradations on a broader scale?

R. Tucker: Yes.

J. O'Connor: I have an offhand comment that in the scientific sense these indications may not even be worth looking for. In a public sense, I think one can say that we probably have quite a number of indications, at least in the Christiaensen Basin and the Upper Hudson Shelf Valley. The only reasonable presumption is that we have less significant impacts on a broader scale than you will ever be able to measure. That is a fair and a public sort of assessment; not the sort of thing one can publish in a journal.

R. Tucker: Fred, you have been out at the 106-mile site. People originally predicted that you would not see effects on the bottom.

F. Grassle: Most of the pollutants that we are concerned about are particle bound, and different size particles behave differently in different systems. We put a lot of effort into modeling our coastal environments, and I think that one of the major new areas that needs to be developed is having adequate models for sediment transport, which includes the surface layers of sediment. But also rates of bioturbation need to be included. How interactive is the bottom with the water column? If we can predict where those particles are going, then we can design our studies of impacts on the environment accordingly. From the point of view of a deep-sea ecologist it is important to follow exactly where those particles were going, particularly since they are a lot like the density of the phytodetritus that gets to and feeds the sea floor. So it is a sort of basic problem similar to phytodetritus, which, only recently, deep-sea ecologists had thought could get to the bottom in measurable quantities in spring blooms. It is possible to track the particles on the sea floor. Now the challenge is to know how they are redistributed. I was very interested in the presentations yesterday and today that suggest that there is a short-term deposition, and that in storms, material gets remobilized and finally is transported further offshore to canyons and the continental slope. Some of the studies that had been done by Cabelli and by Dick Cooper's group on the upper slope environment show widespread distribution of pollutants along that region, especially in the canyons. Tracking those materials on particles is going to be one of the major areas of research in the future. This site is quite interesting because there are a lot of little depositional areas along the continental shelf. There is the mud patch south of Nantucket and similar mud patches around Hatteras as well as the one in the New York Bight apex. They are interesting sites from the standpoint of both reservoirs and sources of pollutants.

R. Tucker: Jack, you raised a question this morning about historical information on fisheries (see Discussion in Wilk et al., this volume). I wonder if you want to expand on the concept of whether we can pick out a general background.

J. Pearce: Well, this morning's brief discussion really had to do with what we know about historical fish populations. We see changes based upon museum collections, earlier data logs, and so on. When I was involved in directing research on ocean dumping, I was narrowly focused on ocean dumping as the biggest issue in the New York Bight. My colleagues, Larry Swanson and Joel O'Connor and others, did a very effective job, through the years, of convincing me that there are many sources of contaminants other than ocean dumping. I believe that is true even more today, particularly having been involved in a number of so-called "blue ribbon panels" or committees having to determine what the principal issues are.

I am also convinced that we must increasingly use generic information from other parts of the world. There are ongoing studies in Finland and in the North Sea having to do with effects of contaminants. I think the issues require more immediate solutions than will be forthcoming from years or decades of additional study, not that more science, i.e. more monitoring, is required, but I think we have to begin to take management steps. People working at the National Center for Atmospheric Research are very concerned about global warming, global climate change; all of these things that worry our Washington administrators and politicians. Although several of these people have recommended additional studies, almost all of them are beginning to say we have got to find solutions now. As I say, ocean dumping is one of many insults to our coastal zone.

Bill Gordon said that he did not think aquaculture would necessarily be a solution to the problem of providing protein. I happen to believe that it will be a significant part of providing more seafood. Human populations are changing their diet very rapidly. Beef is out and fish is in. And yet we are probably fishing our wild stocks to the maximum degree in many instances. It is true, there are some underutilized species; but even if these become used we still will not get much more out of our wild stocks.

If in fact we are to cultivate the seas we have got to manage the coastal zone in a far better way than we have as a nation or even on a worldwide basis. This means that we have got to begin to use generic information, information forthcoming from studies that are done in other places, or use information from the kinds of studies that we have heard over the last two days.

Somebody mentioned that the second biggest cash crop from Maine right now is salmon. A few years ago there was not a salmon coming out of the State of Maine—it was lobsters. Now, it is lobster, salmon, and very quickly, mussels are becoming a new cash crop. We are going to continue to produce protein to provide a gourmet diet, a healthful diet in the United States, and to feed not 10 billion people but, based on the new projections, 11.5 billion people by the year 2030. Now that is a rapid population increase. When I was in China just a year ago, it was obvious how intensively those people have to work their coastal zone and landmass to feed a billion-plus population. The worldwide crisis that we are in has not been generally recognized. We are going to have to use every bit of our coastal zone to provide protein for ourselves and others.

W. Gordon: I want to pick up on a couple of things. Before he died, Ghandi announced his view of the major problem in India. People thought it would be food, or something like that, but his response was a million new Indians a month. And if you remember the recent typhoons in the Orient that killed 160,000 people—that is two weeks of the reproductive rate of replacement. Populations in the coastal zone—where many of those people lived that were killed and where they should not have been in the first place—are cause for grave concern. My point on aquaculture is that if we do not have good, high quality water in the coastal zone we can not have aquaculture. We are coming at one another with a population at the same time we are trying to use that water resource in a beneficial way. If we start aquaculture we have got to deal with the fish waste.

R. Tucker: I understand in Puget Sound they had salmon in pens causing some problems with local water: eutrophication and contamination.

W. Gordon: Several things. One was the visual "pollution" of the fish pens out in front of million dollar houses, but the other part of it was local contamination from the fish waste. We have to deal with those issues, all of which I think have a technical solution. When it comes to population, it is going to be a political decision, and I do not see the will to do that.

R. Tucker: To return to the context of additional sources: Cindy, you mentioned the need to go after the mud dump site as well, but maybe could you expand, from a management point of view, where you see educating citizens or advocacy in terms of addressing some of the other issues.

C. Zipf: I think that a lot can be learned from the sludge experience about the levels of contamination and the solutions to that contamination. The solutions get back to reducing the amount of contaminants in sewage sludge. That works from both a citizen standpoint and an industrial standpoint. I think that you are right; other panelists have suggested that there are technical solutions and I think that is correct. I think what we should stop doing is looking for "quick fix" solutions. At first we started dumping toxic waste onto our land; now we have Superfund sites. We then began discharging wastes into rivers, and we have water quality problems in our rivers—toxic hot spots which generate toxic materials or which ultimately wind up in the sediments. Now we have contaminated sediment problems across the country. We also started putting contaminants into the air and we have acid rain. So "quick fix" technical solutions are not the answer. We have got to get down to the root of the problem and start looking at the environment from the standpoint of what we are generating and the alternatives for those products. It is not acceptable to look for another toxic shell in a toxic shell game. We have got to put a stop to that.

I think that clearly the New Jersey sludge experience has shown us that with the political will, with the citizens behind you and with science verifying the situation, you can make a unique combination, an invincible combination, and push for source reduction. The end of sewage sludge dumping by New Jersey has led to a new sewage sludge policy, one of beneficial reuse. That beneficial reuse policy is leading to reduction of pollutants in the sewage sludge. Not only does that benefit receiving waters from the discharge of sewer treatment plants and help improve the quality of the air from the volatile organics coming from sewage treatment plants, but it improves the whole environmental loop. I think that is the way the decision makers are going to have to go. With the prodding of citizens and scientists, the regulators are going to have to move.

R. Tucker: It is now New Jersey's obligation to find ways to beneficially use sewage sludge. We are in the process of having to look at that issue and I am glad to see that we are moving out of incineration as an alternative to ocean dumping. That is another situation where we are simply diluting it into the general environment.

C. Zipf: I think that from the standpoint of beneficial use, incineration is not considered a beneficial use. It is a reuse of the nutrient value of the sewage sludge that makes beneficial use a viable alternative, not burning it.

R. Tucker: Jim, you talked about stewardship obligations and about a more general approach to our local environment. Do you want to expand on that?

J. Chambers: I was keyed by the use of the term management and what our managers need. Managers are people that are involved in the decision-making processes. They control degradation and all of those topics that I talked about earlier: contaminants cutting off flows of fresh water, wetland losses. Those are the major issues in the judgement of NMFS and NOAA that should be addressed nationwide. In particular, we should try to determine what the contribution of inshore habitats are to the production of offshore fisheries—which ones are critical. And if we lose those, what are we going to be trading off over the long term? Secondly, the effects of man's activities through obliteration, or contamination, cutting off the flows, as a long term effect: I think we ought to be trying to model those long-term effects so that we can predict for policy makers what they are going to be losing with regard to populations in the future. It comes down to pulling all of our multidisciplinary expertise together, which was demonstrated well over the last two days. I think we must capitalize on that kind of multi-attack in scientific effort to answer the major questions that resource managers need and want answered.

W. Gordon: I want to comment on what Jim said. He outlines some needs, but in fact NMFS is not organized to deal with these issues, nor is the Government for that matter. I am struck by the fact that within the Federal level there are 37 different agencies, and nine executive departments, that all have responsibilities in the areas that Jim outlined. The Government's only initiative to deal with this, that I am aware of, is Coastal America, and Congress has effectively cut that out of the budget.

J. Chambers: We are the only agency that is the steward for marine resources and it is our obligation to do it, whether or not Congress wants to put the resources in the steward's hands to effectively carry out that responsibility. We conduct both the research on the effects of man's activities and the importance of habitat. We comment on all of the ten thousand proposed projects that are put forward every year and whether or not they should or should not be built or not built. We do have a central place in that decision-making process and we can affect it. For all manner of Federal project decisions, we have that authority. The problem is whether we have the infantry to carry it out.

R. Tucker: There really are some interrelated questions over the roles of Federal and state agencies and the public. John Keith raised some questions about reality versus perception. But specifically how do we educate the public to see the need to advocate the proper funding for this research? How do we better interact with a wider public? Fred, would you want to address that?

F. Grassle: I do not have any special idea on how best to communicate with the public except that all of us have a responsibility in that area. Cindy's earlier point, that we need to focus our attention on sources, is very important. I agree with that and I think that the land disposal option is going to have a lot of problems in the near future. I also agree with Jack, that the question of ocean disposal is going to come up again; there recently has been some comment in newspapers about that issue. I might mention in the context of this symposium that there was a major meeting in Woods Hole about what ocean disposal might consist of in the future. It is probably worth reviewing at this meeting, just so that you know. There is no reason at this time to consider ocean disposal of waste, but when you look at the population problems of the future, as has been mentioned, it certainly is going to come up again, perhaps not in this country but in other countries. The discussion at Woods Hole asked the question, "If it were to happen, how would one go about doing it"? The conclusion was that any waste disposal in the ocean should be approached in large part the way ocean scientists have been thinking of radioactive waste disposal in the ocean; namely, that you want to get it to the least energetic places, places that have the least productivity, places where there is the least possibility of getting it back—either back to man or having it spread widely in the system. So the conclusions were that low productivity of abyssal plains would be places that one would consider. In those environments anything that was done would be set up as an experiment, where you would actually place materials very precisely on the bottom in an array that was designed by statisticians and you would actually find out what was really going on.

That does not really address or answer your question about communications with the public. I know the Woods Hole symposium was widely touted in the newspapers as saying we should go out and put it back in the ocean and that really was not the result of that meeting.

R. Tucker: I think we have other members of the panel wanting to react to that.

J. Pearce: I am not reacting directly to what Fred said. I will say in regard to education that one panelist earlier brought up the matter that perhaps it is more important to develop appropriate reports and fact sheets that can be disseminated to the general public and used by the managers. I would certainly second that.

One sometimes hears the phrase, "You have got to get your priorities straight." Most scientists that I know get their strokes and even paycheck increases by publishing papers or doing something that catches the attention of the laboratory director or a Washington administrator. You rarely get strokes from writing something that might appear in the *Underwater Naturalist* or the *Audubon Society Magazine* that might reach tens of thousands of people and have some educational impact. Along those lines, yesterday in the *New York Times* science section, an article said that your chances were one in 6,000 of being hit by an asteroid. Being hit by an asteroid turns out to have a greater probability than being struck by lightning, or probably being eaten by a mako shark at Montauk. It is far greater than being killed in an airplane and yet most people get white knuckles every time they get on a 727. Yet in a few weeks, we, as a nation, are now worried about being hit by asteroids.

This is the highest form of "PR" to get additional funding. The astronomers of the world are raising what may be a real issue but also may be a false issue. But the fact is that the astronomers do a very good job of suggesting to the public that there are certain risks out there and they better be sure they pay a bunch of astronomers to look for flying rocks. The next need is Department of Defense money to develop a new atomic pea shooter or something that will be able to divert this thing out of orbit as it is heading for earth. I think we have got to do an equally good job of identifying some of the long-range probabilities that will befall us if we keep living the way we do. That is the highest calling for anyone in this room. You may all think you have a Ph.D. in science that is important and that you have got to publish. The fact is that most of us in this room work for some government agency; we do not work for Rutgers University or the University of Rhode Island to do work that is interesting science. We are paid to do good science that will help the likes of Bill Gordon or Bill Fox or some future administrator solve problems.

In much of what we have discussed, if you were to talk to an administrator and say these are important things we have heard about and we should capitalize on this, the administrator might say, "Forget it, we are going to put ten percent more into counting fish." I think Bill Gordon will verify that. Few people come to Bill Gordon and say, "Solve that ocean dumping problem." They would if they saw more headlines. Congressman Hughes or Howard were people who came and beat upon the administrators and said to deal with this somehow. And probably EPA says, "I hope I can shove this one off to NOAA" and NOAA says, "I hope I can shove this one off to EPA." Part of this is working the power cluster but that is the reality of the world we work in, and we have to learn to play this game to protect fisheries habitats. One of the ways is to begin to find interest groups, the fisherman of the world, who see that their catches are down, and to get them or others, mothers or grandmothers, interested in fisheries habitats. You have got to get people focused on an issue, otherwise, we are wheel spinning.

I would not have believed in 1968 when I started the ocean dumping experience that 23 years later, I would be sitting in a room still talking about ocean dumping. But here we are because we have not played the game right to address the real issues, to get real actions and solutions. I think the solution is in education. We have not figured out how to do this, and we spend too much time fighting between academics, government, and the various interests. We have got to learn how to milk the power clusters.

R. Tucker: I agree it would be much more socially useful if we could take some of that Star Wars technology and point it at asteroids rather than other human populations.

W. Gordon: I would like to pick up on that but first a comment. About three weeks ago on national television, I heard a comment that New Jersey was a terrible place in which to live; that it is overcrowded with people and a filthy place to live. Yet in working here, I now know that New Jersey probably is leading the nation in taking steps to cleanup its environment. The public does not know that. You will hear of New Jersey fresh produce in Chicago, but you will not hear in Chicago that New Jersey is a nice place to visit. The past Governor went on television and tried, with some degree of success, to overcome the debacle of what is washed up on the beaches. I think people are coming back to the coast.

Jack has touched on a good point. We are a rapidly aging population and senior citizens are very concerned. Yet, I dare say there is not a person in this room who has sought out one of the senior citizens or written a popular article describing the situation and what they can do about it. Here is a challenge to anyone in this room to do that; here is a beginning point. The other point is, as scientists, we are not standing up individually or collectively and saying that we have a problem and should begin to deal with it now.

R. Tucker: That is a good point Bill. There is a bill in Congress now which would require more general monitoring along the east coast, essentially using as a model the kind of intensive coastal monitoring New Jersey has been doing. Yet there is a lot of resistance. We are saying that if all of the coastal States monitored as well as we did, then the concern about tourism would not be focused just on New Jersey beaches.

C. Zipf: Just to reflect on the discussion as to how monies can be made available and how to encourage the "grandmothers" activism, I think that it is important that the science community be able to relate their findings more aggressively to the public. We are the ones, the citizen groups, the public interest groups that can do the lobbying during the budgetary process. More than once, Clean Ocean Action has gone to the state and Federal legislatures to lobby for additional funds. Part of our problem has been the hesitancy of the scientific community to err on the side of caution, i.e. we do not really know this and we do not really know that. I think we need ammunition to help provide the funding, whether it is "fish kills from the sky" with respect to atmospheric deposition or some other type of wording. We need to be able to show there are effects in the coastal zone from a multitude of sources.

To touch on the Woods Hole conference, it could be argued that the same lobbying effort of citizens and ocean advocate groups around the country, used to promote funding for looking at effects and helping to improve the ocean condition, could be exerted on those who would wish to put something into the ocean to study it. The ocean advocacy community could be focused on stopping funding for those efforts that would squirt or pump or dump into the ocean environment. The point I tried to raise earlier, with my reflection on some of the myths that were touted not only by politicians but by the educated community, was that "sewage sludge would never reach the bottom at the 106-mile site" or "sewage sludge dumping does not affect the ocean 12 miles offshore." These were all well thought-out statements, but in fact we found them untrue.

We do not know enough about the ocean to consider squirting, pumping, or otherwise dumping in the ocean, and we need to emphasize what we can utilize from the ocean. Such resources can help solve some of the social problems that we have on land through fisheries management or aquaculture and other types of programs.

F. Grassle: I agree with you, Cindy, and radioactive waste disposal is a good example. There is no research being done on that because it is essentially a dead issue to put high level radioactive waste in the ocean. Public advocacy groups are extremely important. In coming to New Jersey, one thing that I have been pleased about is that there is a strong tradition of scientists and public advocacy groups talking to one another. Individual scientists can not really communicate as effectively as the public advocacy groups. The other thing that is important in New Jersey is a tremendous tradition of cooperation of state, Federal and private agencies. Many jurisdictional boundaries are blurred.

And, as for Jack's comment, I think that the main criticism I have heard of academic scientists in New Jersey is that we have not done anything for the fisherman. There are some of those similarities in all of the sciences and it certainly is a national trend that Federal, academic, and state scientists are increasingly going to have to work together. It is a necessary response to a shrinking budget for dealing with these problems.

R. Tucker: I would be very interested in questions or comments from the general audience.

R. Pikanowski: I would like to make two major points. I agree with Jim that habitat research is important. Something you have to remember (I call it the Pikanowski logarithmic scale) is obvious if you look at purely anthropogenic effects on fish stocks for all species combined. Let us say on the east coast, 90% of the fishing mortality comes from fishing, perhaps 9% comes from physical habitat loss, 0.9% from eutrophication, and 0.09% from toxics. So, when you are going to do habitat research you have got to fight that huge confounding signal. It is expensive to do that and something to bear in mind. That does not even account for the natural variations in stocks. If you read Bigelow and Schroeder [1953], fish can do some strange things, like naturally disappear completely for 100 years. I agree we need to do some habitat research but I would bear in mind it can be expensive and has to be carefully thought-out.

As to constituency building, the Sandy Hook staff shrank from 80 to 40, when the building burned, and NOAA did not want us to have a presence here. Jim Howard kept us here. You have got the world's most valuable recreational fisheries and Sandy Hook is right at the focus of it. We once had a few small projects dealing with recreational fisheries, but were asked not to do them anymore. So I am getting very crossed signals, from a total lack of support to suddenly, "Let's get out there, beat the drums, get the money. Come on let's get a thousand people working at the Hook." What is the truth? Which way are we going, or rather, which way can we go?

J. Pearce: What we do have to worry about is something called the world of reality. The first Corps of Engineers [COE] study, the nine volume report that came out in 1969 or 1970, was researched at the huge cost of $200,000, half of the sum Bert Walford took to support the recreational and game fish programs in the lab. I want people to keep things in perspective; $100,000 paid for the first COE study. The next, the famous MESA program, involved something like 30 million dollars, maybe 40 million dollars if you add in other activities. That was a lot of money, but when they were completed we had published numerous reports and volumes, including the one on the New York Bight edited by Garry Mayer (Mayer, 1982). With 30 million invested, they came up with statements and recommendations, about the same as you have heard during this meeting. Today, for the small amount of money that you are talking about, we have produced a better product. In my years of dealing with the New York Bight, I have never seen a better set of statements, with many of them fairly parallel. I do not think money is the problem. What we have got to do is get the word out, not just about the Sandy Hook researchers, but also the academic community.

In many cases we are talking about dealing with larger issues. Nearly 20 years ago, I was focusing on ocean dumping. Ocean dumping is a minor problem today relative to how we use our coastal zones. There is more oil leaking out of automobiles in a collective sense than was spilt in all of the oil spills that have occurred. These facts are significant but on single issues, e.g. ocean dumping or dredge material disposal, we are losing sight of what we do to our environment as a whole. That probably is not what you want to hear but that is the way the world is. If people would address themselves to carrying this kind of information to the general public, we would be far better off.

As an example, there is something going on in Europe that is worth listening to. It is called the precautionary approach. In Europe, in Holland and Scandinavia, they are no longer allowing industries to start any processes discharging new material unless the industry or the community can show that it is not going to harm the environment. Instead of the onus being put on the individuals to fight the power structure, the legislative responsibility is now upon the groups—the institutions—that produce the waste. That is a new way forward which will prevent a lot of deterioration.

Another approach is to base these actions upon historical studies. One of the more interesting examples of this was one commissioned by MESA. The late Tabor Polgar did a historical study of the statistical correlation between dredging release of pollutants with changes in fish populations. I still reference that volume when I am working with the United Nations.

In another instance, Dr. Ken Mountford was at one time very interested in what the anthropologists could tell us about Chesapeake Bay based on Indian middens. From these middens there was evidence of past shellfish species and even fish species that existed in Chesapeake Bay which are no longer found there. We have got to use this kind of information.

W. Gordon: I wanted to add that there are two resources in this room that scientists at Sandy Hook certainly can use. Cindy's organization, Clean Ocean Action, has a tremendous network for publicity. The idea is to reach the public with pertinent information.

R. Tucker: Bob raised a question about fishing pressures confounding measures of habitat degradation. Bill, would you comment about that?

W. Gordon: I agree that fisheries are not managed very well, but fishery management is on the verge of coming of age. I preached this in 1972, and now it is slowly coming into being. It is giving personal property rights to individual fisherman, so he has a respect for the resource. This includes respect for the habitat; he has got something at stake by speaking out when something is not right. I think we will soon see remarkable changes in fisheries management in this country.

We also have to change the attitudes of recreational people. They should be reporting what they catch. Commercial fishermen should be reporting what they catch. We have this centuries-old tradition that it is our God-given right to take what we please. Until we get a handle on fishing mortality we really can not do a good job describing the habitat aspect. There are examples, particularly on the Gulf Coast, where entire estuaries have been bulkheaded or otherwise filled in and the production of shrimp and estuarine-dependent species in that area has gone to zero. We have got to take those examples and say to folks, "Do you want that to happen here?" There is some excellent work that is going to come from Fred Grassle's people, Ken Able in particular, on the production value of the estuaries. It is terribly important that the public is made aware as those results become available. That concept that we are embarking on in conjunction with Anne Studholme and some of her people here at the Northeast Center will also pay off handsomely. The public has to be fed this information rapidly, however, and then you will have a concerned public and one that is reactive.

W. Davis: I think part of the magic of this project was that there was a real invitation for participation. My laboratory and agency [EPA] have benefitted from that invitation. The project grew from hallway discussions with Bob Murchelano and Jack Pearce. The managers invited it and allowed it to grow. Now what I am interested in is some thought regarding where we are going. What is the next project?

R. Tucker: Would anyone on the panel want to address that specific point?

W. Gordon: I only want to comment that as Sea Grant Director I have to pull teeth to get people to work together in the universities and the government agencies, whether it is your agency or National Marine Fisheries Service. I certainly welcome university principle investigators who are willing to collaborate with others, and this last comment was right on. You could move a lot with less if you have people working together.

R. Tucker: I agree completely. We in the State of New Jersey have benefited from working closely with the researchers. We really have been able to get some of the answers to important management questions that we are concerned about. This kind of cooperation is a two way street. The ability to get the kinds of scientific information is one thing but to also get the researchers concerned enough that they can talk in public meetings and raise the consciousness is also extremely important. We have had the privilege of having individuals like Jack and Bill on blue ribbon panels to address some of the development issues and it is extremely important.

S. Clark: I will make a pitch in response to the question that was raised. Where to go next? I would urge NMFS to be more proactive and look at the largest source of pollution in the New York Bight, which is the mud dumpsite. I realize that toxics have a very small effect on fisheries compared to over-fishing. But nevertheless, here is the biggest, largest amount of toxics being dumped into the New York Bight and we do not know the impact. I think that is a big question to answer. I think the past five years was well spent trying to tackle the answer to questions on the effects of sludge dumping at the 12-mile site. Why not turn to something that is ongoing right now?

I would also like to make a comment to NOAA in general: you have all this information about the oceans and fisheries and impacts of anthropogenic inputs, yet the public does not know enough about it. You have to do a better job in marketing that information. For example, this meeting or the report that is coming out of this meeting should have a press conference. Focus not only on New Jersey but New York. Tell people about it so that they know that sludge is not just something that can go back into the ocean, which some people are advocating because they do not want to deal with it on land.

These are really important things that NOAA can play a much bigger role now than it does. Tell Congress. Tell EPA. Get out there. I truly urge NOAA to play a much bigger role in the arena of decision making than it does now.

There are all kinds of programs going on in the New York Harbor aimed at making decisions about source reduction, pollution prevention, managing sewer plants better. Getting a grip on combined sewer outfalls. Where is NOAA? Where is NMFS? Participate and help make these decisions rather than just being on the sidelines.

W. Gordon: I have to respond and add a little story to try to put into perspective NMFS/NOAA's problem. Starting in 1981, I was Director of the National Marine Fisheries Service for six years. During that period of time, I was horrified. Up until then, as a Government employee, I thought we were supposed to be the advocate for the resources and for the habitat. And when I took on the job heading the National Marine Fisheries Service, that is what I thought was part of the job. Let me use an example that I think everybody in this room is aware of—Westway Highway in New York City. We had a fellow stationed in Connecticut who was monitoring that project since its inception, some ten years previously. It came time for decision making and a group in New York brought suit against the Federal Government to prevent the permit from going through. This individual who had been monitoring the project for 10 years gathered all scientific data into a huge document. He was told he could not testify at the trial. So we arranged that he be subpoenaed and therefore had to testify. A Washington bureaucrat went to the trial. When he came back to the public staff meeting (it was not closed to the public but only government officials were present), he reported on the 10 to 11 hours of testimony. He said we have to put out a memorandum that forbids Federal workers from testifying in such circumstances. And I said, "This is not Communism, it is not the Soviet Union."

The decision to start Westway went ahead while I was steadfastly defending the original position. No, it should not be constructed under those circumstances, filling in 240 acres where it was. We were not opposed to Westway Highway per se, simply to the environmental damage it was going to do in the Hudson River. My supervisor said, "We are going to approve it and the fish will move up river."

Now that is the kind of silliness we have encountered since 1980 and which has not totally gone away. The Agency cannot be a flag waving advocacy for the resources or for the habitat in the sense that you are describing. I agree with you, however. That is what I tried to do and was handcuffed in the past.

Just one more anecdote. Another well known scientist, who worked here for a while, was asked to testify by the Congress. That is a command performance; so you write testimony and submit it to the Office of Management and Budget for review. They always modify it because they want to protect the taxpayer from paying more money for this or that. When he testified he was criticized because that was not the "approved" testimony to give. He said, "That is not what I wrote." It was another attempt to muzzle a world-renowned scientist. I respected him for speaking out on the issue; that is what scientists have to do—speak out on issues in a rational way based on facts. If need be, Cindy and I can be a conduit for information. We can arrange for those facts to be made available to the public. Part of the problem is that the government agencies are not flag waving advocates.

C. Zipf: I agree with that, and, quite frankly, I have received more than one unmarked brown envelope in the mail that has provided information we have found useful in bringing the facts to light.

I want to concur with what Sarah Clark said with respect to study of the mud dump. I think it is important because the Christiaensen Basin is really the Interstate 95 used by very important fish species to move into the estuary. This is a crucial habitat area. We do not really know what has happened in the Christiaensen Basin. It would be interesting to dig into the Christiaensen Basin and see to what depths the *Clostridium* counts remain high. We should see how much buildup has occurred of sewage and effluent and plume material in the Christiaensen Basin. I think it is important to maintain a study in that area. We have not decided whether or not mud dumping has an environmental effect.

I would also like to followup on the comment for NOAA to become more proactive. We will attempt to work on our elected officials to get NOAA to be more proactive, but NOAA has the Coastal Zone Management program (CZM) that controls much of the development at the coast. NOAA is not enforcing the CZM restrictions, so development at the shore has bloomed. I think scientists need to provide us with information so that we can be the advocates.

J. Chambers: I want to give some ammunition to those that would use it. We held a 3-day symposium in March [1991], in Baltimore, and invited the best people in the country in various disciplines to define the significance of the situation in terms of what is happening to habitats nationwide, to examine the safety net for protecting those habitats. First I should say that the meeting was sponsored jointly by the commercial fishing community, the recreational fishing community, NOAA, the U.S. Fish and Wildlife Service, and the Chesapeake Bay program. I believe this was the first time that commercial and recreational interests did anything together.

One of the major outcomes was that the commercial fishing industry, as represented by the National Fisheries Institute, has taken on habitat as its next major area of concern. That symposium resulted in an executive summary and solid recommendations including a national policy on habitat; support for the stewardship agency in terms of its stature, its leadership, its resources and people, and finally legislative authority for us to have some teeth, rather than trying to gum the opposition to death as we now do. The Mid-Atlantic

Fisheries Management Council has endorsed the recommendations from that summary and asked each of the other Fisheries Management Councils to do likewise. There are two points to stress: first, there is plenty of information from each of the papers on what sort of trends are occurring in both habitats and resources; second, the recommendations are very solid in terms of what this country needs to do to fix its system so that we can take care of these resources.

D. Gross: My comments have to do with the double-edged sword of public perception. About the article Jack Pearce referred to in the *New York Times,* I do not think the public's attention crystallized around the threat of asteroids until there was a near collision in 1989, yet this is something scientists had known about since perhaps before you began the MESA Project.

The same thing is true, I suppose, with global warming. It took a very warm summer in 1988 to crystallize public, and ultimately Congressional, attention. Yet, the danger with such crystallization is that other events can very quickly erode that support such as a few cold winters or cold summers. Where is such a near collision with regard to water pollution in this area? And is a near calamity what we need or is it going to be something that haunts us rather than helps us ultimately?

Another two-edged sword is that of aquaculture. I agree that aquaculture is probably the wave of many futures. The problem is that often, in the development of aquaculture, there is destruction of coastal habitats and so the balance of having one without the destruction of the other seems to me something we should advocate.

Lastly, there are a growing number of studies, not in this country but in Germany, showing that pollution does have some serious effects on mortality of early life stages of fishes. We are beginning to get a better handle on that here as well.

F. Grassle: I think we can learn from the comment by Jack about the precautionary approach that is taken in Europe on dealing with many of these problems.

J. O'Connor: We could go back in the Apex area to distinguishing sources, one from another. But I have some concern about expending a lot of resources in searching for that sort of Holy Grail as opposed to better characterizing the aggregate effect. Particularly important is to transmit those aggregate effects in simple ways to the public and decision makers.

There was a dogma at one time that unless you knew the source, Government could do nothing about the effect. That idea is being eroded pretty rapidly now and I think we can begin in much broader ways than to look at individual outfall pipes.

One brief comment about education. We should start on the assumption that education is itself a profession, as much a profession as marine science of any kind. It is fine to encourage scientists to write popular articles and fact sheets. I have given up trying to change people's behavior at my age, since I feel most of the scientific community is quite reticent about doing that sort of thing. Perhaps for this activity the more effective approach would be to build in education as part of the continuing activity. Make it a line item in the budget and managers can insure that it gets done as part of the program.

M. Greges: I do not have a question but I have some comments. I was invited to speak on the panel or to be a part of this panel a couple of months ago and when I found out what the topic was, I gracefully declined. The Corps [of Engineers] really has no interest in sludge. We try to separate ourselves as much as possible from sludge; however we suffer from the same misperceptions that your keynote speaker mentioned, that is that dredge material is garbage, that it is hazardous waste, or tampon applicators and hypodermic needles.

Dredge material is sediment and a lot of the results that you get from your tests on sludge have nothing to do with dredge material. I hesitate to even come up here and speak about it but it is been mentioned over the past couple of days and Sarah [Clark] has successfully goaded me up to the microphone with her comments. Cindy had some interesting comments: the solution is dilution.

C. Zipf: Is not dilution.

M. Greges: Well, some people believe that it is dilution, but that has never been New York District's view towards dredge material. That may have been the solution or not the solution for sludge, but dredge material is placed at the mud dumpsite because it is a non-dispersive site. The fact that we have the grotesque warts rising from the ocean floor. (I like that.) The fact that those warts are there is at least a surficial indication that the material is being placed, number one, where it should be placed, and, number two, that it is staying there. You would not have these 40 foot mounds if you had been placing material there year after year and it was being dispersed from that site. I am not saying that there is no loss from those mounds, but there is a good indication that when we say it is a non-dispersive site we have proof.

I wholeheartedly agree with your point about looking at the sources of pollution problems. I think that is the priority study or investigation that we should be involved with, which leads me to Sarah's [Clark] comment about the mud dumpsite being, "the biggest source of pollution in the Bight." We have a different contention. We believe that we are relocating material—some of it polluted, some of it contaminated—which is there already. It is the point and non-point source pollution sources that you should be going after. Dredge material itself is not the pollutant. It is not the contaminant. The contaminants are already in there. What we are doing is taking it out of channels to protect safe navigation and putting it in an area where we believe it will stay put.

Do we have to do more work in monitoring? Do we have to do further investigations as to how much material we actually lose and determine where the material goes? Yes, I agree, and I think you know this because we have discussed it before. The only thing we have not concluded or agreed on is what type of tests to do, how much the investigation will cost, and exactly what it will show. We should be looking into that, but I would like to shy away from the perception that it is the dredge material that is contributing to the pollution.

Returning to perception, there were a couple of misperceptions about dredge material that I heard at this conference. Yesterday there was a good paper presented with one conclusion that dioxin found at site A–41 was probably due to previously disposed Newark Bay dredge material at the mud dump, as well as from natural sedimentation processes and from the 12-mile site (see Bopp et al., this volume). That one woke me up. I spoke with the author of that paper and he agreed it would be very difficult to prove that that dioxin was from the mud dump. I postulated that it could have been from an errant barge load of material that perhaps was disposed of in the wrong spot. This he agreed with. At the concluding portion of the program on the overhead was the statement inferring dioxin found at A–41 was due to the dredge material site. Again the negative perception was that there was some investigative proof that the dredge site was the source. Yet the author admitted it would be very hard to prove that that was the source of dioxin.

Later we were treated to a graph entitled "Pollutant Loadings"; there is a good title (see O'Reilly, this volume). It had five or six pie charts most of which had a nice big chunk attributing pollutant loadings to dredge material and a nice little sliver to sludge material. Again, this was another surprise. Speaking with the presenter I was a little confused when he said, "Well that is just the pollutant loadings into the Bight, including what goes to the mud dump." The impression that I got was that dredge material is the major contributor to contaminants in the entire Bight. It probably would have been worth mentioning that those contaminants are disposed of in those grotesque warts, or what I like to call "discrete mounds," and that that material was already in the system. We did not go out and add pollutants into the Bight. Again, a very negative misperception.

This morning's presentation on seabed oxygen consumption also tried to make a correlation relating the dredge material dumpsite and oxygen consumption. Yet when someone asked a question the presenter admitted that there was no good data that correlates dredge material to seabed oxygen consumption. Again, for a group of scientists and managers you should be as sensitive to misperceptions about dredge material as to the misperceptions the public has about sludge. There are enough real problems with disposal of dredge material, with monitoring that disposal, and with the contaminants in the material itself. We do not need to be battling innuendo or comments where there are no scientific data or investigations to support it. The most important thing to look at is the sources of pollution and once you stop this, you will stop the contaminants and pollutants found in dredge material.

R. Tucker: I think the COE is certainly to be commended for sponsoring and funding the 1970's study. Until Jack gave some of the history, I did not realize that it was only $100,000 that actually went for that part of the study. I would suggest that may be a good model for the COE to contribute to studies on the actual impacts of the mud dumpsite.

C. Zipf: It is all a matter of perception, whether you see them as mounds or grotesque warts. I think that science indicates there is not enough evidence to discuss what is happening with the dredge material. Some of the most interesting work in the beginning of the symposium was the effect of physical aspects in the Christiaensen Basin area. I never would have thought that wind patterns, etc., would have been effective down deep in the Christiaensen Basin, in the "Mud Hole," so to speak. The effects these patterns have on a 45-foot mound is an open question. I think that the environmental impact statements that have been done on the mud dump are old and need to be revisited to see the range of effects of the mud dump. What is happening to that mound? Is it a mound of course sands that have been collected over time since 1888 or is it light fluff material, with the more contaminated fraction floating into the Christiaensen Basin? Such questions still need to be addressed.

J. Pearce: The comments in this discussion will also provide an answer again to Dan Gross who asked, "Where is that issue that will focus the public's attention?"

I mentioned a while back that we had to work with generic information, wherever it might come from. While it has been lost in history, the Sandy Hook Laboratory invested some $1.2 million of Navy money into the famous Thames River study, which was a study of ocean disposal of dredge material. This was an interagency study, involving EPA, the U.S. Navy, the Sandy Hook Laboratory, and the University of Connecticut. Investigators included Frank Bolan, a physical oceanographer, and the late Sung Feng, probably one of the better marine physiologists. After some years, I kept hoping there would be an aberration that would somehow show a lot wrong. Then we would have a big problem that could focus the attention of the public and get another million or two of Navy money. Unfortunately that did not happen. It turned out the lobsters liked the big huge wart on the floor of Long Island Sound, better than they did the normal flat sandy flats. As I review scores of dredge material studies, inevitably I come up with similar findings.

Feng and I both felt that whatever materials that were there of a toxic nature, by and large, are lost at the dredge site. If anyone has ever seen either a bucket dredge or a suction dredge in operation you will know that the nephloid layer, that layer of ooze containing most of the contaminants, is put in suspension in the water mass in the Thames River, or the Hudson River, or any place where dredging is going on. That material is carried seaward. Unfortunately, this is the price to be paid for dredging to keep the U.S. harbors open. You will find that anything contrary to this is going to fall on deaf public ears. I can not think of a single politician or administrator who would choose to tie up shipping in New York Harbor until such time as we resolve this question.

Coming back to Dan Gross, we must define the problem on which to focus the public's attention. Public attention has to be focused on things like dog feces, which wash off the streets of New York into the Hudson River and add to the pollutant burden. Public attention has to be focused on vehicles that leak oil and produce black smoke. These contribute to pollution burdens that later have to be dredged up and taken somewhere else. Until the public is willing to have vehicles inspected twice a year and are willing to clean up after their pets, there will be bacteria, trace metals, and petroleum hydrocarbons from nonpoint sources working their way into the Hudson River, Newark Bay, Thames River estuary, Buzzard's Bay, and wherever you may be. You may talk it to death and point at the "warts" on the sea floor but the fact is that it is individual activities that create marine pollution, and we have to focus public attention on this issue.

Now, Dan Gross also talked about aquaculture. He talked about German studies, which demonstrated effects on larval fish. Aaron Rosenthal did some of the best laboratory studies I have seen relating various forms of contaminants to malformations of fish larvae. Later, Volkert Dethlefsen made a super case out of dumping certain materials in the German Bight. Dethlefsen spoke at major national hearings, but, in fact, never could really back up his findings. This has become somewhat of an embarrassment to true environmentalists who want to manage the marine environment. It points out how a person, thinking he has a cause celebre, can create a bigger problem in terms of managing the marine environment when he does not have his facts straight. There are real problems however, and we ought to find the sources. Competent marine scientists not only want to know the sources, they want to know the fates. They want to be able to model the fates, quantitate what happens when some contaminant is coming out of a combined sewer outfall or somebody's Buick.

Finally, there is no use having sources and fates unless there are effects. You have got to demonstrate effects. Thus I come full circle to the Thames River study. We were unable to demonstrate any effects even though the Pfizer Corporation, the Electric Boat Company, and others have been putting contaminants into the Thames River for years.

C. Zipf: From a public standpoint, there is a tremendous surge of interest in the environment. The citizenry is interested in what they can do to help reduce ocean pollution. At Clean Ocean Action, we call it "pointless" pollution since the term "nonpoint source" sometimes gets confusing. We have a program of stenciling little blue fish on storm drains, combined with an aggressive citizen education program, to make sure that every community member knows that storm drains lead directly to where fish live. It is not that the citizenry has been ignored—the citizenry is being educated. We can not resist this opportunity of inviting all these scientists to focus on areas where the citizens really do not have a lot of impact, and one is dredge dumping.

More and more, we should show the economic impact of these environmental effects. Not until we had floatable incidents on beaches, where there was a tremendous economic impact, did we find a lot of opportunity for action. So tying the environmental degradation to economics is a way in which we have been able to implement change.

T. Bigford: I have a couple of comments. The first goes back to Bob Pikanowski's exchange with Jim Chambers. When we talk about mortality and the need to address some of the habitat issues that might be part of the total fishery management equation, I do not think we could look at just the mortality that Bob was talking about. Maybe 90% is fishing related but there are a lot of other issues involved—physiological change, behavioral changes, and others. We can not look just at that issue based on fishing mortality. There may be many things happening that change the recruitment that are more subtle less than just a mortality question.

The person mentioned from Milford, Connecticut, who worked on the Westway project is Mike Ludwig.

There are 15 Mike Ludwigs in the Northeast Region who desperately need the scientific and political type of support that I have seen the last two days. Mike and his colleagues, and I am one of them, work on habitat issues that are as far offshore as the 12-mile dumpsite or nearer the coast. I would like to see a bit more effort focused on these issues and not just activities that affect mortality. What is decreasing the health of species, their physiological responses, their behavior?—things other than just whether they are alive or dead.

One closing comment: I have liked what I have heard the last two days including Jack's comment that we should look overseas to what is happening around the world. It is incumbent on people who have been doing this research to think about how their work might apply elsewhere. Instead of letting other people look at your reports and determine how it applies to a disposal site off the mouth of Chesapeake Bay or in Boston Harbor, the scientists most familiar with the data should interpret the information; determine how you can stretch the experimental design and still make decent assessments. You should look at your results and give serious thought to what it means to the general issues of ocean disposal and sewage sludge handling. That is your job and I think you should do it as part of this project. If not all, at least some of you. As you know, the general trusteeship responsibilities fall to the National Marine Fisheries Service.

M. Kaplan: Do you at Sandy Hook or in any of the NMFS groups have a public outreach program? If not, we in New Jersey have found over the last several months, especially with sludge management, that the most effective technique is to bring the public into the process. Regardless of what the issue is, we have citizen advisory groups.

I know as scientists we like to do our work and we do not want to be bothered with having to go to public meetings. Presumably, we have better things to do. Because you have a decentralized regional system with your labs at Sandy Hook, Woods Hole, etc., it seems like you have a perfect opportunity to know the people on a local level and to bring them into the process. Let them know what research is going on, what your legislators need to know, and how the whole system can be integrated. It works and it does not just have to be citizens but all members of the community including industry. If you want to identify source reduction you must point to every single person responsible for generating sludge. Bob [Tucker] and I would be happy to help. We have a communications group in our program in Science and Research and it works. You could learn a lot from it if you do not have one of your own.

R. Tucker: That certainly gets back to the point that the education has to be interactive.

J. Chambers: I will tell you the reality. The answer is that we do not have any outreach. We are trying to do it on our own hook as amateurs. In fact, for our resource base at two of our major laboratories, we provide base funds: in one case 29% of their operating budget, in the other case 35%. This is to conduct research on the effects of organic contaminants for the importance of wetlands to living resources. We can only pay for a third of the operations; the rest they have to beg, borrow and steal every year. That is the reality of the situation. We are not able to afford any outreach type organization, a situation that has existed for the 14 years since I have been there.

W. Gordon: NOAA has an outreach program, known as the Marine Advisory Service. It is funded through Sea Grant in cooperation with the cooperative extension operations in some States; sometimes on their own, sometimes with county support. Since 1980, this entire Sea Grant program, until a couple of years ago, was zeroed out at the Administration's budget request to Congress. Last year the core research programs were recommended for funding at 25 million dollars, but the Marine Advisory Service, at a requested rate of about 15 million dollars, was recommended for zero. Clearly, NOAA, the Department of Commerce, and the Administration, really does not care about outreach. It is the Congress that adds it back.

R. Tucker: We have raised some important points. I know we can go on but it is time to end. I would like particularly to thank the audience for their participation. This panel has been extremely interesting and responsive to the questions asked. The whole two days have been tremendous. I would finally like to really thank Anne and her staff at Sandy Hook and all of the other people who are responsible for putting on this tremendous program.

Closing Remarks

R. Murchelano: I sense some real trepidation as to what the next move will be. I could tether you here to hear more statements that may prove controversial to you but I prefer not to do this. After hearing some of the comments from Bill and Cindy I could utilize this panel very effectively, in terms of the go-between aspects. We will need your help in disseminating information which we may not wish to promote ourselves.

Let me address that point for a moment. We do have statements prepared by the staff at Sandy Hook that cover all the issues that were addressed at this meeting. I'd like to see them in the *New York Times,* so Jack can quote them next week; or the *Washington Post,* the *New-*

ark Star Ledger, or *Philadelphia Inquirer*. That is the challenge if you wish to disseminate information that you heard here today and be our surrogate for getting the information before the media. I certainly would like to thank everyone again for attending.

Finally, I agree with our leadoff speaker from Sandy Hook, Bob Pikanowski. The greatest cause of mortality in fish stocks is certainly fishing mortality. It then becomes a question of how much does someone, arbitrarily, either intelligently or politically, spend in other areas, that is on habitat or environmental work. I happen to feel that the distribution of funds at present is inadequate for doing the other side of what is part of fisheries research. Perhaps some of that can be achieved by better definition, better focus, defining more practical problems, or more realistic priorities. The allocation between what is given for stock assessment activities in the agency and habitat or environmental work is too imbalanced. There are better ways to do some of the things which are needed for conservation and utilization of marine resources. If, as was mentioned, these things were rocks, we would not be interested, but they are not rocks. We eat them; they are there as a sustainable resource. It is renewable and there are other things that we can do and should do.

Thank you all for coming. We certainly enjoyed the experience. I know all of us at Sandy Hook, Woods Hole, and Narragansett, did, and we look forward to the publication of a volume which you can critique.

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EXECUTIVE SUMMARY

At the initiation of the 12-mile study in 1986, no single issue related to the use of coastal waters in the New York Bight had generated more public concern than ocean disposal of waste material, particularly sewage sludge. The public's perception was that sludge dumping at the 12-mile site was a major factor affecting water quality and the utilization of marine resources. At the time, predictions of the effects of dumping cessation ranged from no significant or measurable improvement (New York City Department of Environmental Protection[1]) to modest changes including reduction in pathogenic bacteria and viruses in sediments over a limited geographical area (Swanson et al., 1985), with any large-scale improvements resulting only from comparable reductions in disposal of dredged material or other wastes (Gunnerson et al., 1982).

For three years, intensive field studies focused on three specific areas: 1) the replicate stations, NY6, R2, and NY11, which represented habitats that were respectively heavily degraded, enriched, and less severely influenced by sludge dumping; 2) the broadscale stations located throughout a 350 km^2 area in the New York Bight apex, which provided a more descriptive background for the study; and 3) the Hudson Shelf Valley, in order to determine the fate of contaminants dispersed from the site.

The major conclusions resulting from the study demonstrated that some indicators of sewage pollution in sediments, such as the rate of oxygen consumption by the seabed, redox potential, bottom dissolved-oxygen gradients, total bacteria count, and counts of coliform bacteria, responded rapidly to diminishing inputs of sewage sludge. Measurements of lead enrichment, *Clostridium perfringens* spores, and redox potential demonstrated a decrease in sewage contamination over a large area surrounding the dumpsite, but only in surficial sediments. The responses of the benthic community to the cessation of sludge dumping were not as apparent. Decreases in pollution-tolerant benthic species and increases in the number of species and the number of crustaceans were documented at the most heavily polluted site, but the distribution of megafauna captured in trawls showed no appreciable changes that could be attributed to cessation of dumping.

Hydrography

During the study period, hydrographic conditions fell within the normal range for the inner New York Bight, and in general, water column variability in temperature and salinity occurred at all time scales.

Current meters, deployed one meter above the bottom from May 1987 through June 1989, indicated that currents at subtidal frequencies were highly coherent with wind, especially at sites within the Hudson Shelf Valley. During the summer, the most intense bottom flow events (36 cm/s maximum) were in a southward (down-valley) direction driven by occasional episodes of westward winds. During the winter, persistent eastward winds would often become strong enough to sustain bottom flow in a northward (up-valley) direction (54 cm/s maximum).

Repeated hydrographic surveys indicated an exchange of bottom water in the dumpsite area over one week, associated with flow events up and down the Hudson Shelf Valley.

Sediment transport was also event dominated, depending on the coincidence of strong bottom currents and wave action to provide resuspension energy. In the upper Hudson Shelf Valley, sediment resuspension occurs approximately 5% of the year, primarily during winter months.

Dissolved Oxygen and Bottom Oxygen Gradients

The role played by sediments in the Christiaensen Basin in annual episodes of hypoxia in the New York Bight apex has decreased following abatement of sewage sludge dumping as suggested by various changes such as:

1. disappearance of high rates of seabed oxygen consumption in the Basin,
2. disappearance of strong near-bottom gradients in dissolved oxygen,
3. decrease in the area of strongly reducing surface sediments (redox), and
4. decreases in acid-soluble sulfide in surface sediments.

In 1989, dissolved oxygen (DO) concentrations in water near the seabed reached 2.5 mg/l, the lowest value observed since reductions in sludge dumping were begun in 1986. At the most heavily degraded area

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[1] New York City Department of Environmental Protection. 1983. Technical information to support the redesignation of the 12-mile site for the ocean disposal of municipal sewage sludge. Prepared by Ecological Analysts, Inc. Sparks, MD, and SEAMOcean, Inc. Wheaton, MD. New York City Department of Environmental Protection, 2358 Municipal Bldg., New York, NY 10007. Unpubl. manuscr. 483 p. + appendices.

(NY6), values below 0.5 mg/l were observed in summer months during 1983-1985. From 1986 to 1988, dissolved oxygen minima at this station did not fall below about 4 mg/l. The 1989 value was within the range predicted for the study and probably reflects a general decline of DO levels throughout the apex due to water column processes unrelated to sludge dumping.

Seabed Oxygen Consumption (SOC)

Consumption of oxygen by the seabed is used as a measure of benthic community metabolism to understand energy flow and carbon cycling at the sediment-water interface in aquatic ecosystems.

Rates of SOC were responsive to increases and decreases in carbon loading to the Christiaensen Basin. SOC increased 57% in the sludge-affected area between 1974 and 1983 when the amount of sludge dumped had increased 89%. During dumping, rates of SOC at NY6 (Station 33) were highly elevated during summer (60 ml $0_2/m^2 \bullet h$) and were high even when bottom temperatures were low. Following cessation, SOC rates decreased to the background levels (15-20 ml $0_2/m^2 \bullet h$) recorded in the New York Bight at stations unaffected by dumping and responding minimally to seasonal fluctuations in bottom water temperature.

These observed changes were in good agreement with the anticipated changes hypothesized prior to the study. Station 30, 2.3 km east of the 12-mile site and outside the Christiaensen Basin, exhibited no significant response to changes in the quantity of sludge dumped. Seabed oxygen consumption rates at Station 35, adjacent to the 6-mile site, responded more to changes in dredge inputs than to changes in sludge loading.

Changes in Sediment Biogeochemistry

Two years after cessation of sludge dumping, the concentration of biologically labile carbon in the sediment at the most heavily degraded station (NY6) had decreased to one-third the level observed during dumping, consistent with predictions from a model of sediment biogeochemistry. Concentrations were still higher than at the enriched (R2) and reference (NY11) stations; the difference between the latter two was attributable to differential input of phytoplankton carbon.

In response to this decrease, redox potential in surface sediments at the degraded and enriched stations generally increased and seasonal fluctuations were diminished. The highest redox values were recorded in 1989, with values at all stations beginning to converge, also consistent with model predictions. The observed increase in the annual minimum redox in surface sediments at NY6 is approximately double that hypothesized at the beginning of the study.

During dumping, highly reducing conditions were observed in surface sediments throughout the Christiaensen Basin during the summer. Approximately 200 km^2 of the 350 km^2 survey area had redox values <50 mV in surface sediments. Approximately two years after dumping ceased, only 15 km^2 had values <50 mV.

A significant portion of the temporal variation in the number of benthic macrofaunal species was correlated with sediment redox potential.

Sediment Metal Contamination

Statistically significant decreases in lead and lead enrichment (ratio of present lead/iron to pre-industrial lead/iron) were found at the most heavily degraded station (NY6), occurring within one month following the cessation of dumping. Other metal concentrations also decreased to about one-half to one-fourth the concentrations observed during dumping, which was not the order of magnitude decrease hypothesized prior to the study. No significant temporal changes were observed 4–5 cm below the sediment surface.

There was, however, a marked decrease in the area with lead enrichment >20 (indicating heavy sludge pollution) from 105 km^2 during dumping to 25 km^2 only 1.7 years after cessation. It was estimated that within four years these highly contaminated surface sediments would disappear.

Sediment Grain Size and Total Organic Carbon

Analysis of grain size and total organic carbon (TOC) indicated that NY6 showed a significant response to the cessation of dumping. Prior to cessation, sediment contained a greater silt fraction and had higher porosities, lower bulk density, and higher TOC than R2 and NY11; sediment porosity decreased significantly from mean pre-cessation levels of 68% to 61% after dumping stopped. TOC decreased from 4–5% to around 2%; bulk density increased from 1.4 g/cm^3 to 1.6 g/cm^3, due to rapid mineralization of organic matter and dispersion of fine sludge.

Sediment Organic Contamination

Organic contaminants, i.e. PAH's, PCB's, and DDT-related pesticides, in sediments were highest at the more heavily degraded station NY6, less at enriched station R2, and least at the reference station NY11.

Mean concentrations were lower (though not statistically significant) in 1989 than 1986 during dumping.

Sediment Deposition and Resuspension

Measurement of fallout (Cesium-137) and natural (Berylium-7) radionuclides in sediment cores indicated that the highest rates of net accumulation of fine-grained particles (a few centimeters per year) in the study area occurred along the axis of the Hudson Shelf Valley up to several kilometers down-valley from the dredged spoil disposal site and former sewage sludge disposal site. This suggests resuspension of disposed material and down-valley transport.

The high concentration of 2,3,7,8-TCDD (dioxin) found in sediment samples between the sludge and dredged material dumpsites probably resulted from the disposal of materials dredged from the lower Passaic River and Newark Bay.

Erodibility and sludge marker studies along the Hudson Shelf Valley indicated short-term storage of sludge in the Christiaensen Basin followed by resuspension events probably coinciding with wind-driven currents. An exponential down-valley decline in sediment erodibility and sludge markers (chemical contaminants and *Clostridium perfringens* spores) was demonstrated, but station depth and width of the Hudson Shelf Valley governed deposition and subsequent resuspension potential. A predictable relationship between grain size and erodibility was demonstrated, permitting development of erosion models based on shear, grain size, and porosity values.

The peak in sediment erodibility shifted down the Hudson Shelf Valley about 50 km during 1987–89 except for an unexpected peak that appeared in the Christiaensen Basin-Hudson Shelf Valley in 1989. The unexpected increase in erodibility and *Clostridium* spores in June 1989 may be explained by the increase in dumping of dredged material at the 6-mile site during that spring. It is also likely that heavy metals found in upper Hudson Shelf Valley sediments resulted from the dredged materials disposed at that site rather than from the 12-mile dumpsite.

The dominance of Dichlorodiphenyltrichlorethane (DDT) over Dichlorodiphenyldichlorethane (DDD) in surface particulate matter samples indicates that resuspension and Hudson-Raritan discharge were only minor sources of these compounds to the surface waters. Based on analysis of samples from the shelf break, it is conjectured that the DDT contamination of surface waters in the vicinity of the dumpsites is dominated by regional inputs derived from coastal aerosol transport during spraying of DDT prior to the domestic ban in 1972. The dominance of DDD over DDT in the near-bottom, suspended-particle sample is evidence of resuspension.

Abundance and Distribution of *Clostridium perfringens*

During dumping, exceptionally high concentrations of *C. perfringens* spores (1.7×10^5/g), a specific indicator of sewage contamination, were observed at station NY6, the most polluted station nearest the dumpsite. Spore concentrations increased with increasing bottom depth throughout the study area and decreased with distance from the 12-mile dumpsite. Significant decreases in spore concentrations, temporally related to the cessation of dumping, were observed at this and two other sites sampled frequently. During dumping, approximately 27% of the 350 km^2 area surveyed had *C. perfringens* spore concentrations exceeding 1×10^4/g; 3.3 years following cessation, only 10% of the area surpassed these levels.

In April 1991, sediment profiles revealed generally increasing densities of *Clostridium* spores with sediment depth. Spore concentrations 2–10 cm below the surface were comparable to concentrations observed in the surface layer during dumping. Also spore concentrations in surface sediments at NY6 were still exceptionally high ($>1 \times 10^4$/g) and levels comparable to New York Bight "background" (10–100 spores/g) would not be expected until the year 2000.

Demersal Finfish and Megainvertebrates

There was considerable spatial homogeneity in the distribution and abundance of species captured in trawls. After cessation of dumping, there were no significant changes in distribution and abundance that could not be accounted for by natural variability, except for a 60% increase in lobster abundance at the most heavily degraded station, NY6. Lobster fishing effort increased in the vicinity of the dumpsite largely because of the elimination of pot fouling by sludge.

Returns from winter flounder, *Pleuronectes americanus*, tagged between July 1986 and August 1989 indicated that fish moved between the dumpsite area and the embayments of New York and New Jersey as well as into more northerly waters. Winter flounder from the dumpsite may represent a mix of inshore populations that may be in transit from areas that become suboptimal or may be part of the spawning migration.

From observations of gross pathology of winter flounder, the incidence of finrot and internal lesions were apparently reduced, supporting the initial working hypothesis that disease would decrease. This low level reflects the same pattern of decline in finrot observed in the New York Bight over the past two decades. There were no significant changes in incidence of lymphocystis (viral) or *Glugea* (microsporidial) infections or in genetic abnormalities (bentfin and ambicoloration).

There were only minor shifts in the diets of winter flounder, red hake, *Urophycis chuss*, and American lobster, *Homarus americanus*, after the cessation of dumping or among the sludge-influenced stations. This may be due to the species' mobility and high degree of variability in food habits. The notable exception to this was the decrease in frequency of *Capitella capitata* (a pollutant-indicator polychaete) in winter flounder stomachs at the most heavily degraded station (NY6) from 20% during dumping to less than 0.1% in the post-dumping period. This reflects the decrease in *Capitella* populations at that site. In addition, sewage artifacts (hair balls, fibers) in lobster and red hake stomachs decreased markedly, signaling improvement in the quality of ingested material and presumably forage habitat.

Analyses of organic contaminants in hepatic tissues of winter flounder and lobster collected from the vicinity of the dumpsite during dumping and after cessation showed that concentrations of PCB's in both species were significantly higher in samples collected from nearer the dumpsite than from the reference area. In addition, lobster values were significantly higher than those for winter flounder.

Benthic Species

As predicted prior to the study, significant increases in the number of crustaceans, molluscs, and total species collected in grab samples occurred at the most heavily degraded station (NY6), and increases were greater at both sludge-influenced stations (NY6, R2) than at the reference area (NY11). The decline in the pollutant-indicator species *Capitella capitata* coupled with the reappearance of more pollution sensitive amphipods further supported the chemical evidence that habitat quality had improved within two years after dumping stopped. However, samples collected 38 months after dumping ended still contained quantities of material identified as sludge related.

Shellfish Closure Area

After cessation of dumping, numbers of coliform bacteria decreased substantially in the sediments in and around the dumpsite but remained high in deeper waters to the west and down the Hudson Shelf Valley. Although *Clostridium* spores continued to persist, as of May 1990 most of the areas sampled, with the exception of the Christiaensen Basin, would have met the current standards to permit reopening of the shellfish growing area. This would allow the U.S. Food and Drug Administration to reassess closure areas and reopen the area for harvesting.

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